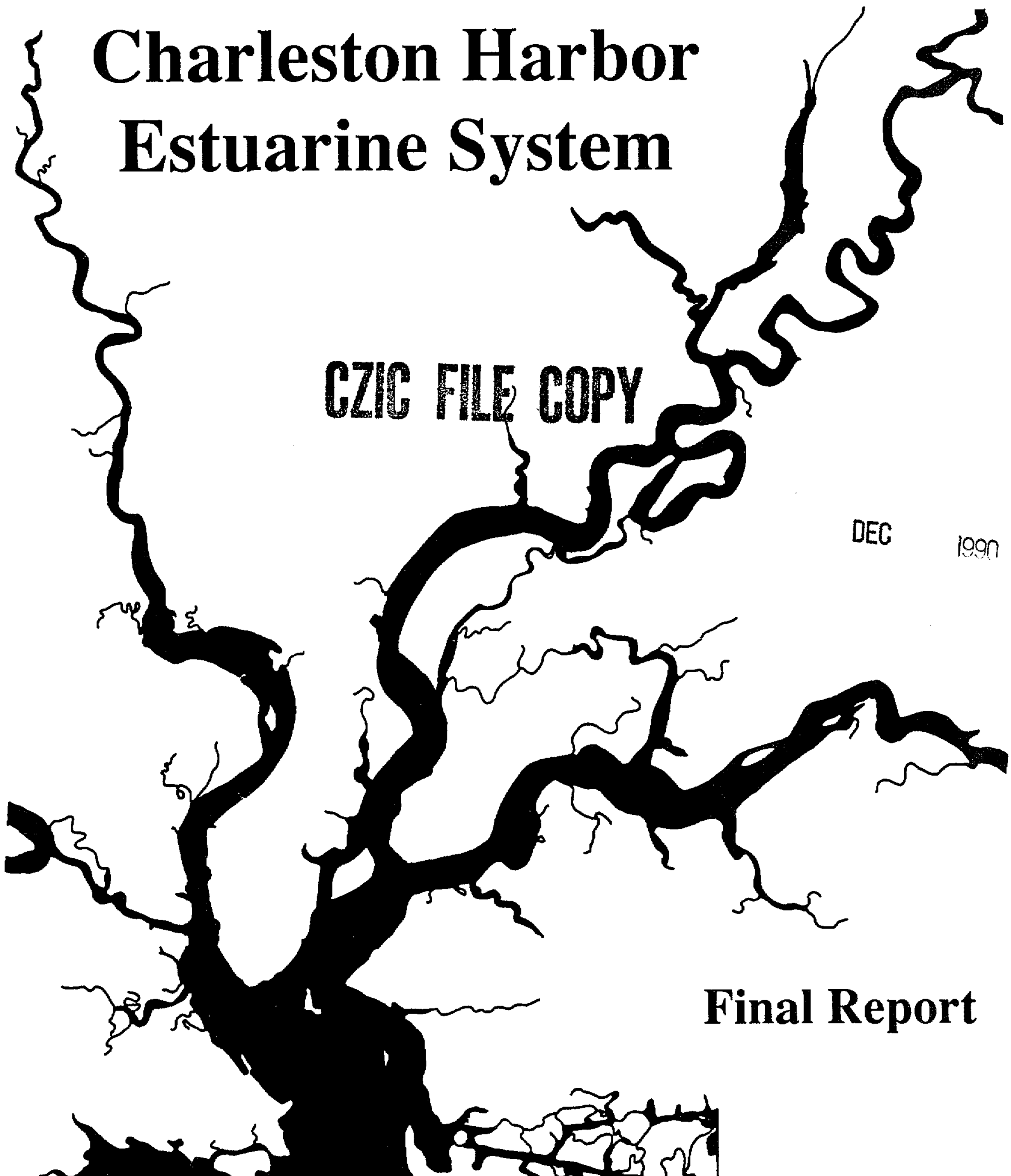
A Physical and Ecological Characterization of the Charleston Harbor Estuarine System



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Charleston, South Carolina

A PHYSICAL AND ECOLOGICAL CHARACTERIZATION OF THE CHARLESTON HARBOR ESTUARINE SYSTEM

Edited by

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Submitted by
the

Marine Resources Division
South Carolina Wildlife and Marine Resources Department

in cooperation with
the

South Carolina Sea Grant Consortium
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University of South Carolina
and
The Citadel

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TABLE OF CONTENTS

Page

CHAPTER I - Introduction 1

Background 1
Study Objectives 4
Study Participants 5

CHAPTER II - Description of the Study Area 7

CHAPTER III - Hydrography 11

Introduction 11
Methods 12
Results and Discussion 17
Freshwater Flow 17
Temperature 18
Salinity 20
Dissolved Oxygen 30
Turbidity 38
Nutrients 41
Summary 45

CHAPTER IV - Organic Carbon and Nutrient Dynamics 47

Introduction 47
Natural Patterns of Estuarine Nutrient Distribution 47
Goals and Objectives 49
Methods 50
Field Sampling 50
Chemical Analyses 52
Statistical Analyses 53
Results and Discussion 53
River Flow and Salinity Distributions 53
Turbidity and Dissolved Oxygen 56
Organic Carbon 63
Inorganic Nutrients 78
Summary 97

CHAPTER V - Physical Dynamics 99

Introduction 99
Background and Objectives 99
Review of the Diversion and Rediversion Projects 100

TABLE OF CONTENTS: Continued

Methods	102
Longitudinal Salinity Measurements	104
Cross-Sectional Measurements	105
Velocity, Tide, and Flow Measurements	105
Data Reduction	105
Numerical Modeling: Circulation and Dispersion	108
Time-Varying Boundary Formulation	112
Results and Discussion	112
Summary	120
CHAPTER VI - Sediments	121
Introduction	121
Methods	124
Results and Discussion	127
Seasonal Study	127
Short-Term Intensive Study	135
Summary	146
CHAPTER VII - Benthic Macrofaunal Communities	149
Introduction	149
Long-Term Seasonal Study	150
Methods	150
Results and Discussion	152
Cooper River	152
Wando River	169
Charleston Harbor	181
Ashley River	194
Synopsis of Long-Term Spatial and Temporal Patterns	209
Short-Term Intensive Study	212
Methods	212
Field and Laboratory	212
Data Analysis	214
Results and Discussion	215
Community Diversity and Abundance Patterns	215
Dominant Taxa	222
General Community Composition and Distribution Patterns	224
Distribution of Benthic Assemblages in Relation to Anthropogenic Activities	235
Summary	237

TABLE OF CONTENTS: Continued

Page

CHAPTER VIII - Finfish and Invertebrate Communities 241

Introduction 241
Methods 242
Data Collection 242
Statistical Analyses 244
Results and Discussion 245
Environmental Parameters 245
Catch Analysis 245
Community Pattern Analysis 260
Species Observations 268
Community Analysis 272
Summary 285

CHAPTER IX - Recruitment of Fishes and Decapod Crustaceans Into Nursery Habitats 289

Introduction 289
Study Sites 291
Settlement of Brachyuran Megalopae 291
Methods 291
Field and Laboratory Procedures 291
Data Analyses 294
Results and Discussion 295
Physical and Meteorological Data 295
Species Composition and Temporal Abundance 299
Diel Variation 312
Lunar Effects 315
Decapod Crustaceans and Fishes from Tidal Marsh Creeks 317
Methods 317
Field and Laboratory Procedures 317
Data Analysis 319
Results and Discussion 320
Physical Characterization of Sites 320
Species Composition and Community Structure 322
Recruitment Patterns 329
Adult Stocks of Blue Crab 364
Methods 364
Results and Discussion 365
Summary 381

TABLE OF CONTENTS: Continued <u>Page</u>

CHAPTER X - Macrophyte Vegetation of Freshwater Tidal Marsh in the Cooper River 385

Introduction 385
Methods 386
Site Description 386
Field Sampling 388
Results and Discussion 388
Summary 401

CHAPTER XI - Contaminants 403

Introduction 403
Methods 404
Sediments 404
Tissue 408
Results and Discussion 409
Metals 414
Organics 422
Summary 424

CHAPTER XII - Technical Workshops 425

Introduction 425
Methods 426
Results 427
Hydrography, Circulation, and Modeling Workshop 427
Water Quality, Nutrient Fluxes, and Contaminants Workshop 428
Estuarine Fauna Workshop 430
Phytoplankton and Macrophyte Productivity Workshop 433
Comments on Planning and Data Management. 436
Discussion 436
Summary 440

LITERATURE CITED 443

APPENDIX 477

CHAPTER I

INTRODUCTION

by

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BACKGROUND

Charleston Harbor encompasses the lower portion of a large and important estuarine system in the southeastern United States. It is located in the central portion of South Carolina's coastline and is formed by the confluence of the Cooper, Ashley and Wando Rivers (Figure I.1). Together, these rivers comprise more than 26,000 hectares of valuable coastal marshlands and open-water habitat, and they form the third largest estuarine drainage area in the state (Tiner, 1977; NOAA, 1985). The wide variety of habitats present in the estuary support a diverse array of flora and fauna, including more than 80 species of macrophytes, 580 planktonic taxa and over 570 macroinvertebrate and finfish species (Van Dolah and Davis, 1989; Davis and Van Dolah, 1990).

Many of the more abundant demersal finfish and crustacean species in the estuary are economically valuable. The harbor system supports large populations of white shrimp (*Penaeus setiferus*), brown shrimp (*P. aztecus*) and blue crab (*Callinectes sapidus*) which are harvested both commercially and recreationally. Although none of the finfish species are commercially harvested within the estuary, many are recreationally important, such as red drum (*Sciaenops ocellata*), spotted sea trout (*Cynoscion nebulosus*), flounder (*Paralichthys lethostigma, P. dentatus*), spot (*Leiostomus xanthurus*), Atlantic croaker (*Micropogonius undulatus*) and catfish (*Ictalurus catus, I. furcatus*). These species and most of the other flora and fauna in the estuary have been influenced to some degree by a variety of anthropogenic activities which also make this estuary important in the southeast.

Charleston Harbor's port facilities represent the largest economic resource associated with the estuary. During 1989, more than 1300 commercial vessels passed through the port and the combined cargo handled in that year exceeded 6.9 million tons, making Charleston Harbor the second largest container port along the Atlantic seaboard. Additionally, the US Navy maintains its third largest home port in the Cooper River. This facility supports more

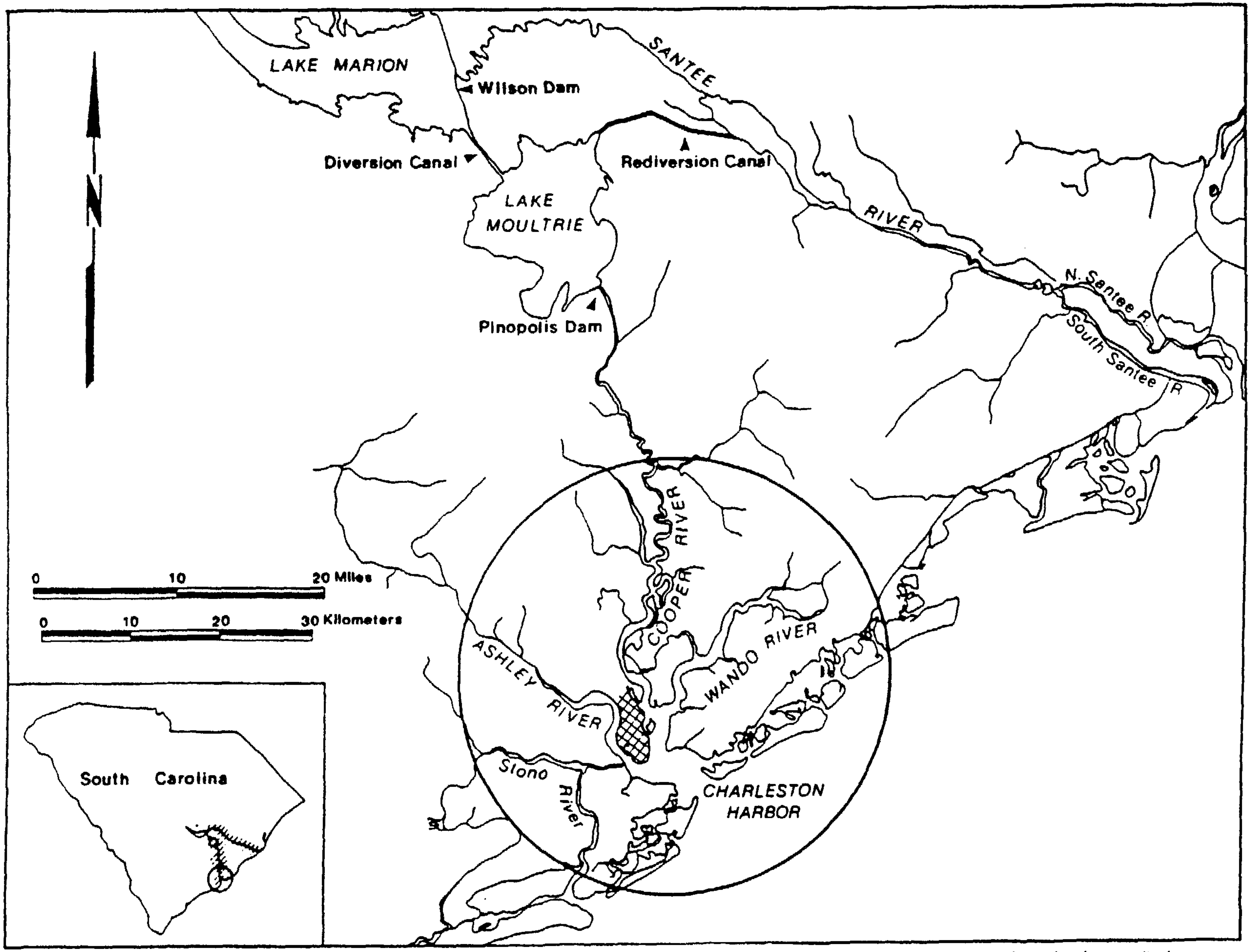

Figure I.1. Map of the lower Santee-Cooper drainage system. The landward boundary of the circle encompassing Charleston Harbor defines the approximate estuarine limits in the major river systems forming the Charleston Harbor estuary.

than 70 surface vessels and submarines, as well as a shipyard and naval weapons station. Both the commercial and naval port facilities have required extensive dredging for maintenance and deepening of the shipping channels in recent years.

Much of the sedimentation in Charleston Harbor prior to 1985 has been attributed to a major water diversion project which was completed by the South Carolina Public Service Authority (SCPCA) in 1942. This project involved construction of the Wilson Dam on the Santee River to form Lake Marion, construction of the Pinopolis Dam at the headwaters of the Cooper River to form Lake Moultrie, and construction of a canal between the two lakes through which approximately 88% of the freshwater flow from the Santee River was directed to the Cooper River (Figure I.1; Little, 1974a; Kjerfve, 1976; Kjerfve and Magill, 1990; USACOE, 1975). This change increased the fresh water flow into Cooper River to approximately 442 m^3/s (Kjerfve, 1976).

To alleviate the shoaling problems attributed to the 1942 diversion project, the US Army Corps of Engineers (USACOE) initiated the "Cooper River Rediversion Project", which was completed in August of 1985. This project rediverted approximately 70% of the water flow from the Cooper River back into the Santee River through a canal in the vicinity of St. Stephens, South Carolina (Figure I.1). Since rediversion, the monthly mean flow into the Pinopolis Dam has been reduced to approximately 122 m^3/s. (see Chapter V for additional details on the diversion and rediversion projects.)

Major hydrographic changes expected in the estuary as a result of the Cooper River Rediversion Project were: 1) an extension of the estuarine boundaries through increased saltwater intrusion, 2) redistribution of salinity regimes within the estuary, 3) a change in the hydraulic character of the harbor from a stratified to a vertically mixed estuary, 4) changes in the current patterns within the estuary, 5) reduction of water levels in the upper Cooper River, and 6) changes in the dilution and flushing rates of pollutants in the system (USACOE, 1975; Benson, 1976, 1977). These hydrographic changes would obviously influence the ecological characteristics of the estuary as well. For example, salinity shifts within the estuary would affect the distribution of many plant and animal species and may have long-term effects on the overall abundance of some species. The degree of distributional changes would be dependent on the degree of salinity changes in different portions of the estuary and the species tolerance or affinity for particular salinity regimes during different stages of their life cycles (Wenner et al., 1984; Van Dolah et al., 1989). Unfortunately, many of these changes may never be well understood due to a lack of sufficient pre-rediversion data for most of the floral and faunal constitutients. Some of the changes may also involve several years of successional change before the communities stabilize to post-rediversion conditions. Even though the full extent of changes resulting from the rediversion project may not be well defined, a comprehensive data base on the physical and biological conditions within the estuary following rediversion was clearly needed.

STUDY OBJECTIVES

The studies described in this report were part of a comprehensive survey program designed to better document hydrographic conditions and selected biological communities in the estuarine portions of the Cooper River, Wando River, Ashley River and Charleston Harbor basin following rediversion. Objectives of the major study components were to:

1) Describe hydrographic conditions in the above river systems following rediversion, and identify seasonal changes in basic water quality parameters throughout these estuaries.

2) Conduct additional studies to characterize the organic carbon and nutrient dynamics and the physical dynamics of the estuary following rediversion.

3) Describe seasonal and yearly changes in macrobenthic infaunal communities present along the estuarine gradient in each of the above river systems, and evaluate the distribution of these communities in relation to various natural and anthropogenic environmental parameters. These studies would also provide information on the distribution of surficial sediments within the estuary.

4) Describe seasonal and yearly changes in the fish and crustacean communities present along the estuarine gradient in each of the above river systems, with particular emphasis on (a) defining the distribution of recreationally and commercially important species of finfish, shrimp and crabs, and (b) identifying patterns of recruitment for some of these species in different portions of the estuary.

5) Describe changes in the macrophyte communities in the upper Cooper River where changes in water levels were expected.

6) Document organic and inorganic contaminant concentrations throughout each of the above estuaries, with particular emphasis on determining the pollutant levels in sediments and the tissue of important fish, crustacean, and mollusk species.

7) Evaluate the significance of changes resulting from rediversion by comparing the post-rediversion data base with data obtained prior to rediversion, where available.

8) Convene a series of research/technical workshops in order to identify and evaluate extant information/data sources, with particular emphasis on (a) identifying major data needs and strategies for acquiring needed research data, and (b) determining the requirements and mechanics for establishing a comprehensive information/data base for the estuary.

STUDY PARTICIPANTS

Researchers from several institutions participated in this study. These institutions included the Marine Resources Division of the South Carolina Wildlife and Marine Resources Department, the South Carolina Sea Grant Consortium, the University of South Carolina, The Citadel, and the South Carolina Department of Health and Environmental Control. Principal investigators associated with each of the study components are identified as authors in the following chapters. Many of these individuals also contributed significantly to several of the study components. Additional study participants include the following: Drs. V. Burrell and P. Sandifer served as co-principal investigators for the overall study effort along with R. Van Dolah, the project coordinator. Ms. M. Davidson and Mr. R. Devoe administered study efforts conducted by Sea Grant as well as the research conducted by USC and Citadel personnel. Other administrative staff who assisted significantly include D. Gibson and J. Groves. Technical support for field and laboratory work was provided by J. Althausen, R. Devlin, V. Greene, M. Katuna, D. Knott, S. Miller, J. Morris, and G. Phillips, J. Sneed, E. Somody, S. Stonehill, and several laboratory staff at DHEC. M. Katuna, C. Wenner, M. Goodwin, and R. DeVoe also contributed to reviews of some chapters. Captain V. Taylor piloted the research vessel *Anita* for most of the trawling efforts. Computer processing support was provided by A. Jahnke, M. Clise, W. Coon, and K. Davis and word processing support was largely provided by M. Lentz with assistance by L. Greene, and M. Bannon. M. Lentz and M. Bannon also contributed significantly to the preparation of the final report. Ms. K. Swanson assisted in graphics preparation for SCWMRD staff.

CHAPTER II

DESCRIPTION OF THE STUDY AREA

by

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The studies described in the following chapters primarily involved sampling in the harbor basin and the estuarine extent of each river system. This represents only a small portion of the entire watershed, which is the second largest in South Carolina and encompasses a drainage area of 41,000 km^2 (NOAA, 1985). A brief history of the changes in the watershed over the past 50 years is described in Chapter I. This chapter provides a description of the present characteristics of the estuary; including adjacent land use patterns.

The Charleston Harbor basin covers an area of 65 km^2 and drains an additional 104 km^2 of local marsh and lowlands. The average depth of the harbor basin at mean low water (MLW) is 3.7 m, and navigation channels are currently being deepened to a depth of 12.2 m (USACOE, 1958, 1966a, 1975). Charleston Harbor's mean tidal range is approximately 1.6 m, spring tides average 1.9 m, and the highest astronomical tides exceed 2.1 m (USDOC, 1989). For the studies described in this report, the mouth of the harbor was considered to be at the junction of the two main channel reaches near Fort Sumter (Figure II.1). Estuarine waters actually extend seaward of this point into the Charleston Harbor Plume with the position of the plume varying dependent on tide stage.

Barrier islands and jetties form the entrance of Charleston Harbor, and most of the basin is surrounded by city and urban developments. As noted previously, the harbor receives considerable shipping traffic due to the large commercial and U.S. Navy port facilities which are located in the harbor basin as well as in the Cooper and Wando Rivers. The commercial port facilities in the basin are located on the Charleston City peninsula (Figure II.1). The Atlantic Intracoastal Waterway (AIWW) also crosses the harbor basin between Mt. Pleasant and James Island at Wappoo Creek. Although there are no major industries located in the lower harbor system, the basin receives effluents from two large sewage treatment facilities which provide secondary wastewater treatment. These are located on Plum Island, and on Mt. Pleasant near the AIWW. Other sources of pollution affecting the lower harbor include non-point source runoff from the city and urban areas,

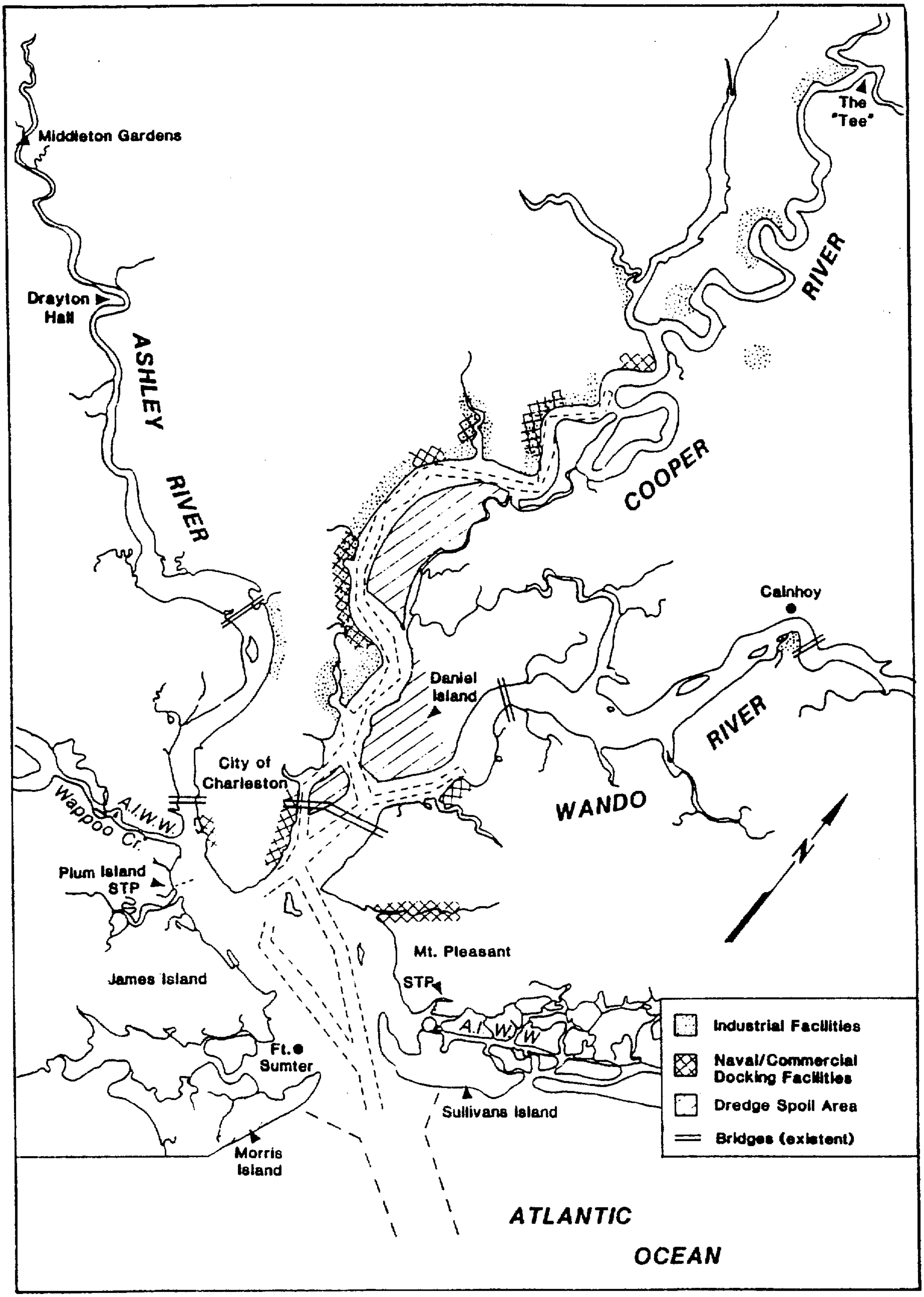

Figure II.1. Map of the Charleston Harbor estuary showing key landmarks and some adjacent land use patterns.

several marina facilities near the mouth of the Ashley River, and runoff and discharges from numerous sources in the three river systems (see below). Several diked disposal areas for dredged materials are located in the harbor basin, with the largest being Drum Island. Water quality in the harbor is rated as SC by the South Carolina Department of Health and Environmental Control (SCDHEC). This rating applies to tidal saltwaters suitable for secondary contact recreation, crabbing, and fishing, except for the harvesting of clams, mussels, or oysters for market purposes or consumption (SCDHEC, 1985). Waters rated as SC should not have dissolved oxygen concentrations less than 4 mg/l and fecal coliform concentrations should not exceed a geometric mean of 1000 colonies/100 ml based on five consecutive samples taken within a 30 day period (see SCDHEC, 1985 for additional conditions). Although these concentrations have been exceeded occasionally, recent reviews of data collected by SCDHEC indicate that water quality within the harbor basin often meets SB standards for dissolved oxygen and fecal coliform levels (Chestnut, 1989; Davis and Van Dolah, 1990). An SB rating applies to tidal salt waters suitable for primary contact recreation, and having a daily average of dissolved oxygen not less than 5 mg/l with fecal coliform concentrations not exceeding a geometric mean of 200/100 ml based on 5 consecutive samples during a 30 day period (see SCDHEC, 1985 for further conditions).

The Cooper River drainage basin is extremely complex due to the construction of the Santee-Cooper Hydroelectric project, but can be divided into three distinct components: the area downstream of the Pinopolis Dam; the area above the Pinopolis Dam, including Lake Moultrie, the diversion canal and Lake Marion; and the upper Santee River drainage basin which extends approximately 400 km from the headwaters of the Santee River drainage basin in the western North Carolina Blue Ridge Mountains. Only the tidal component located below the Pinopolis Dam will be considered in this report and sampling was limited primarily to the estuarine portion which now extends approximately 50 km upriver from the mouth of the harbor to an area near the junction of the East and West Branches of the Cooper River (Figure II.1)

The Cooper River has the greatest concentration of industrial and port facilities among the three river systems forming the Charleston Harbor estuary (Figure II.1). The majority of these are located on the western shoreline and include the U.S. Navy port facilities, commercial facilities associated with the State Ports Authority, and private companies. To accommodate the ship traffic, a 10.7 m deep navigation channel is maintained in the lower Cooper River extending 32 km upstream from the mouth of the river (USACOE, 1966b, 1975). The eastern shoreline of the Cooper River is largely undeveloped, although there are several large diked disposal areas along the length of the maintained channel. The water quality rating for the Cooper River is SC throughout the study area.

The Ashley River flows approximately 50 km from its headwaters in Cypress Swamp in Berkeley county to its junction with the Intracoastal Waterway on the south side of the Charleston Peninsula, where it empties into the lower harbor basin (Figure II.1) (Little, 1974a; Mathews et al., 1980). The river basin drains a 900 km^2 area of marsh and lowlands, spread out over Berkeley and Charleston Counties (Little, 1974a). Depths of the natural channel in the river range from 1.8 m to 11.0 m, and are influenced by tidal action throughout the river's length (USACOE, 1958). The estuarine limits in this river extend approximately 40 km from the mouth of the harbor to an area above Middleton Gardens. The Ashley River has the second largest number of industrial and commercial facilities which are located on the eastern shoreline. Much of the remaining upland areas on both sides of the river support residential developments. Water quality in the Ashley River is rated SC throughout the study area.

The Wando River flows approximately 38 km from its headwaters in Iron Swamp in Charleston County, to its junction with the Cooper River on the north side of the Charleston Peninsula (Figure II.1). The river basin drains a 310 km^2 area of marsh and lowlands, and its depth ranges from 1.5 m to 12.8 m within its natural channel (USACOE, 1957; SCWRC, 1973, 1975). The Wando River is influenced by tidal action throughout its entire length and estuarine waters extend into the creeks which form the upper limits of this river. This river presently has the least upland development compared to the other two river systems, except in its lower reaches. In that area, the State Ports Authority maintains the Wando Terminal Facility which is located on the eastern shoreline. There are also several residential communities which are either already present or being developed on this shoreline. Large diked disposal areas are located on Daniel Island, which forms the western shoreline of the Wando River. The only major industrial facility on this river is Detyens Shipyard located at Cainhoy (Figure II.1) Water quality in the Wando River was recently upgraded to SA above the Wando Terminal. This rating applies to tidal salt waters suitable for harvesting of clams, mussels, or oysters for human consumption. SA waters must maintain a daily dissolved oxygen concentration of 5 mg/l or higher and have median coliform concentrations of 70 colonies/100 ml or less (SCDHEC, 1985). Water quality in the lower portion of the Wando River is rated SB.

Additional descriptions of the physical and biological characteristics in various portions of the Charleston Harbor estuary are provided in the following chapters, along with descriptions of specific sites sampled. Davis and Van Dolah (1990) also provide an extensive description of the pre-rediversion characteristics of this estuary as well as a summary of data sources collected prior to 1985.

CHAPTER III

HYDROGRAPHY

by

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INTRODUCTION

The Charleston Harbor estuary is a complex system with respect to its physical and chemical properties. Hydrographic circulation patterns, sedimentation patterns and the distribution of basic water quality parameters, such as temperature, salinity, dissolved oxygen, pH and nutrients are all strongly affected by climatic conditions, tides, and freshwater flow. As described in Chapter I, freshwater flow has varied considerably over the past 50 years due to major water control projects. Briefly, these included the original diversion project completed in 1942, which increased the flow of freshwater into the harbor to a monthly average of 442 m^3/s, with seasonal variation which ranged from 87 to 844 m^3/s (USACOE, 1966; Kjerfve and Magill, 1990). The long term daily average flow into the harbor between 1942 and 1984 was estimated by NOAA (1985) to be 496 m^3/s. According to Kjerfve and Magill (1990), the monthly average Cooper River flow was 418 m^3/s when evaporation was considered. Freshwater flow was normally highest during the winter months (January - March) and lowest during the autumn months (September - November) during the period of diversion, with spring floods resulting in greater flows (SCWRC, 1979; NOAA, 1985). In 1985 the USACOE rediverted much of the flow back into the Santee Rivers, which resulted in the reduction of freshwater flow in the Cooper River to a low, relatively stable average of 122 m^3/s.

Prior to rediversion, a number of studies were conducted to evaluate basic water quality in the Charleston Harbor estuary. These studies collected hydrographic data such as temperature, salinity, dissolved oxygen, nutrients and turbidity, and have been reviewed by Davis and Van Dolah (1990). The majority of these data sources involved only short-term sampling conducted for specific research projects or environmental impact statements, and were limited to small areas of the estuary (Davis and Van Dolah, 1990). Two data sources, however, contain long-term, basic hydrographic data collected from stations located throughout the estuary. The more comprehensive database is that of the South Carolina Department of Health and Environmental Control (Chestnut, 1989). This database contains hydrographic data from numerous stations throughout the estuary prior to rediversion (Davis and Van Dolah, 1990), 25 of which were sampled for more than 3 years during the period 1970-1985. The second database was collected by the SCWMRD during the Estuarine Survey Study between 1973 and 1978 (Mathews and Shealy, 1978, 1982). The Estuarine Survey Study collected hydrographic data on a quarterly basis from 9 stations in the Charleston Harbor estuary including 3 in the harbor basin, 4 in the Cooper River and 1 in each of the Ashley and Wando Rivers.

Several other studies collected hydrographic data from specific areas of the estuary including: the Wando River by Enwright Laboratories, Inc. (1984) between 1980 and 1984; the upper Cooper River by the United States Geological Survey (unpublished) during 1970-1973 and 1978-1985; the mid Cooper River during the Cooper River Environmental Study (Nelson, 1974) during 1973; and the Wando River during the Wando River Environmental Study (SCWRC, 1974) during 1972.

Rediversion of the Cooper River in 1985 was expected to markedly affect some of the physical and chemical properties of the estuary, while other properties would remain relatively unchanged. For example, water temperatures were not expected to be affected by rediversion, whereas salinity regimes in the Cooper River were expected to change considerably. Studies reported in this chapter were initiated by SCWMRD in 1984 to document the changes in physical and chemical parameters brought about by rediversion. Sampling for this study was completed in December, 1988.

METHODS

Hydrographic data were collected during trawl, grab and hydrographic sampling periods in Charleston Harbor and its tributaries between November, 1984 and December, 1988. The sampling frequency for individual sites and parameters varied from semiannual to monthly during this time period as summarized in Figure III.1. Quarterly hydrographic sampling transects were initiated on the Cooper River in May, 1985, and were designed to follow low slack tide up the river from the mouth of Charleston Harbor to the Tee (Figure III.2). Hydrographic transects were extended to the Ashley and Wando Rivers in October, 1987, and were sampled at both high and low slack tides on a monthly basis through December, 1988. A total of 37 hydrographic stations were sampled during each of two tidal stages per month during this period. These stations included: 16 stations extending 53.8

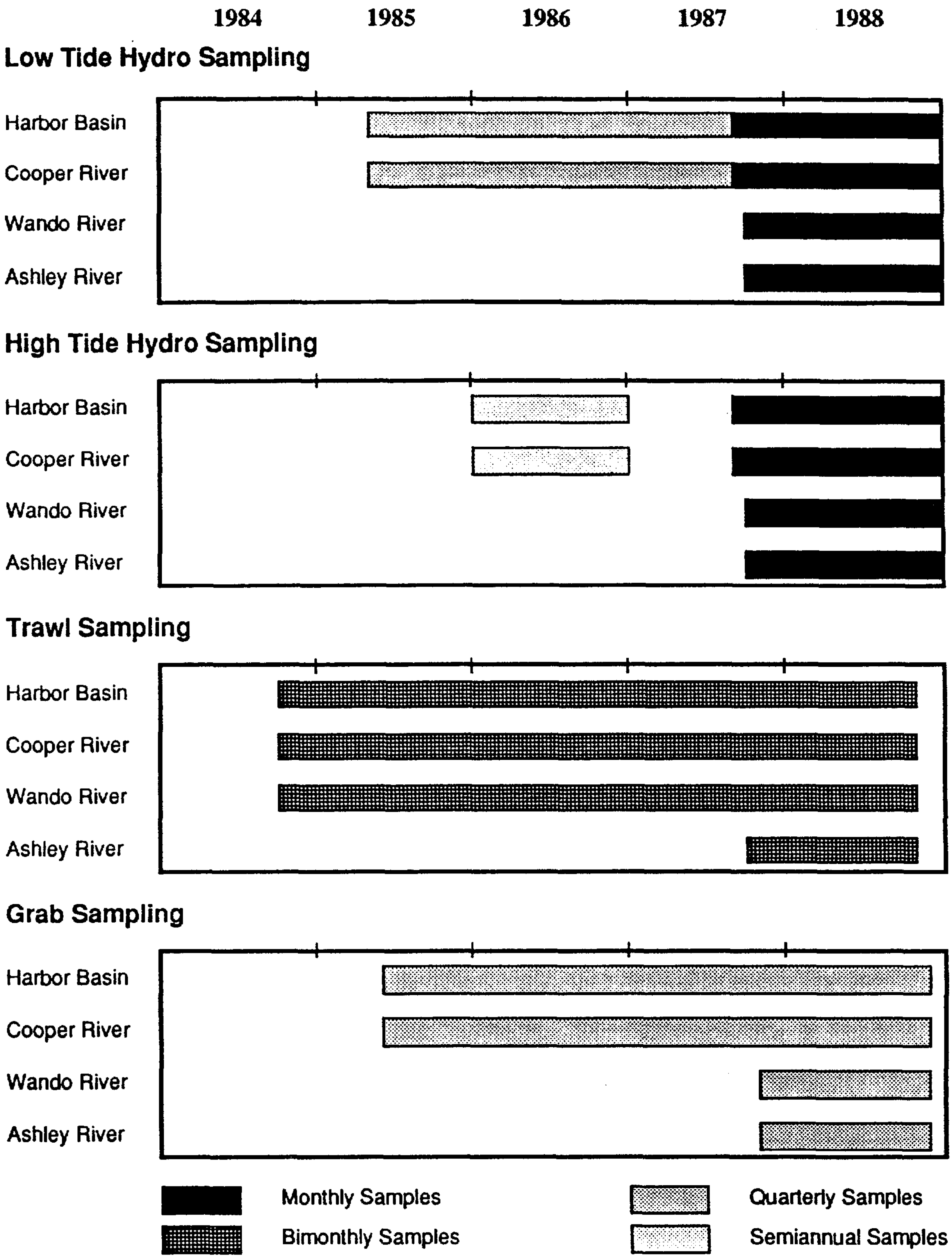

Figure III.1 Timelines depicting periods and frequencies of trawl, grab, and high and low tide hydrographic sampling for each basin in the Charleston Harbor estuary.

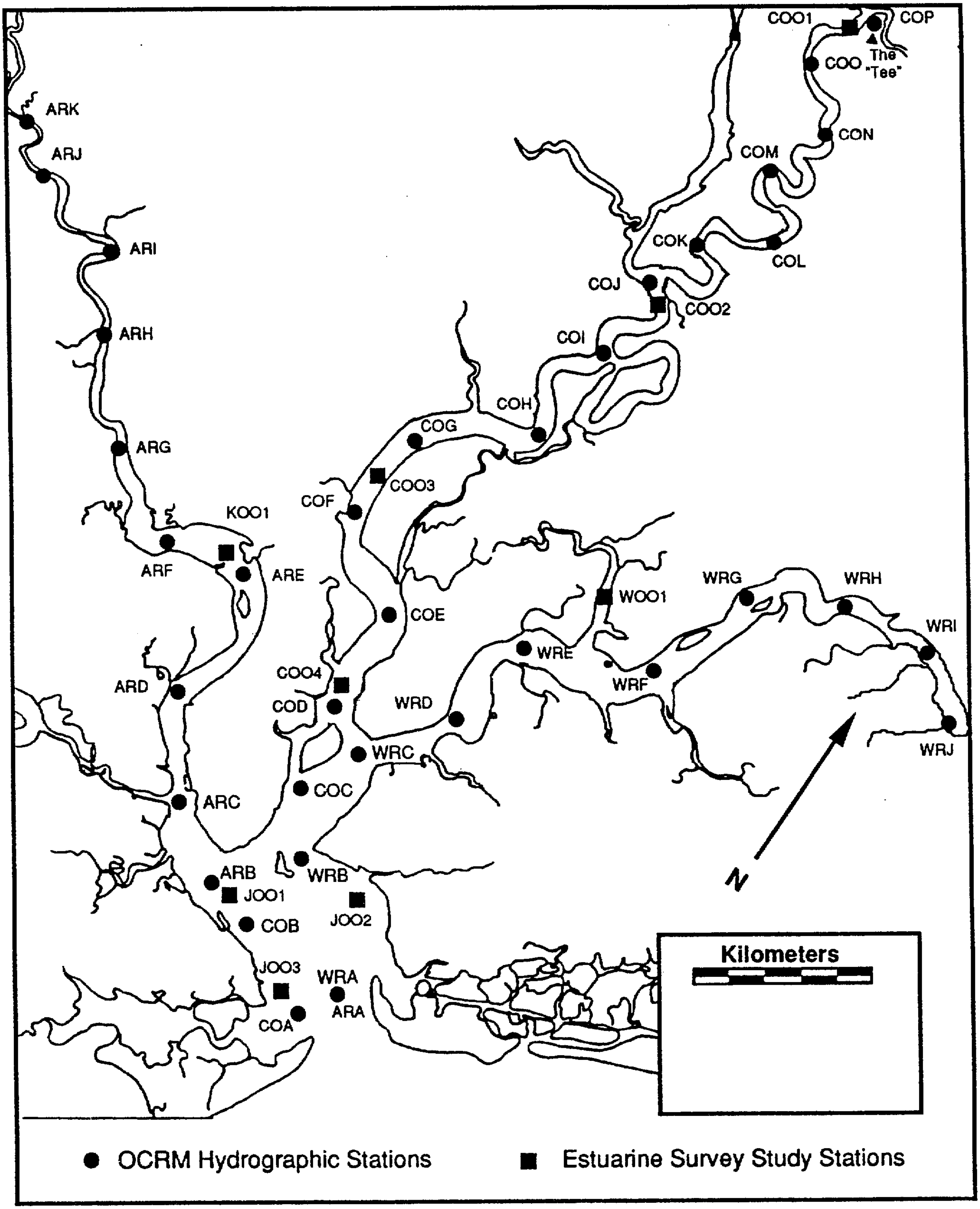

Figure III.2. Location of hydrographic stations sampled during this study and the Estuarine Survey Study hydrographic stations.

km up the Cooper River; 10 stations extending 34.4 km up the Wando River; and 11 stations extending 37.0 km up the Ashley River. In addition, surface and bottom hydrographic data were generally collected during quarterly grab sampling at 13 sites and bimonthly trawl sampling at 12 sites throughout the survey period (Figure III.3). Surface and bottom hydrographic data were also collected on a single occasion at each of 178 sites sampled during an extensive study of benthic communities and bottom sediment characteristics conducted in July, 1988.

Hydrographic parameters were measured *in situ* using a Hydrolab Environmental Data Systems Model SVR2-SU Sonde Unit (Hydrolab, Inc., Austin, Texas). The parameters measured included specific conductance, salinity, temperature, depth and dissolved oxygen. Surface data were collected at a depth of approximately 0.5 m, while bottom data were collected at approximately 0.5 m above the bottom. The Hydrolab unit was calibrated prior to each sampling period, and recalibrated upon completion of sampling. The approximate resolutions and calibration accuracies for the parameters measured by the Hydrolab unit are presented in Appendix III.1. The percent saturation for dissolved oxygen was calculated based on the Unit Standard Atmospheric Concentration by volume as described by Benson and Krause (1984), using the salinity and temperature data for each sample.

Surface and bottom water samples were collected monthly for nutrient determinations at 15 stations throughout the estuary during the low tide transect runs conducted from January, 1988 to December, 1988. Samples were collected with a 1-liter Kemmerer bottle at surface (0.5 m) and bottom (0.5 m off the bottom) depths. Sample aliquots (250-ml) were stored on ice until they could be filtered in the laboratory (within 6 hours). Filtered samples were stored at 4°C, and analyzed as soon as possible. Samples were analyzed for nitrates, nitrites and ortho-phosphates utilizing USEPA methods 352.1, 354.1 and 365.3 respectively (USEPA, 1981a) on a Milton Roy Spectronic 501 ultraviolet-visible spectrophotometer. The spectrophotometer was calibrated over a wide range of values for all three nutrients utilizing 15 standard samples at the beginning of nutrient analysis, and again 6 months later. Each set of samples was further calibrated to the standardization curve utilizing replicate blank samples and standards. In addition, samples were analyzed for total ammonia at the University of South Carolina on a Technicon Autoanalyzer II.

Surface and bottom turbidity samples were collected at the 15 nutrient stations bimonthly between January, 1988 and December, 1988 during the low tide hydrographic transects. Samples were preserved with a few drops of mercuric chloride, and stored at 4°C until they could be analyzed using a calibrated Hach Model 2100A Turbidimeter.

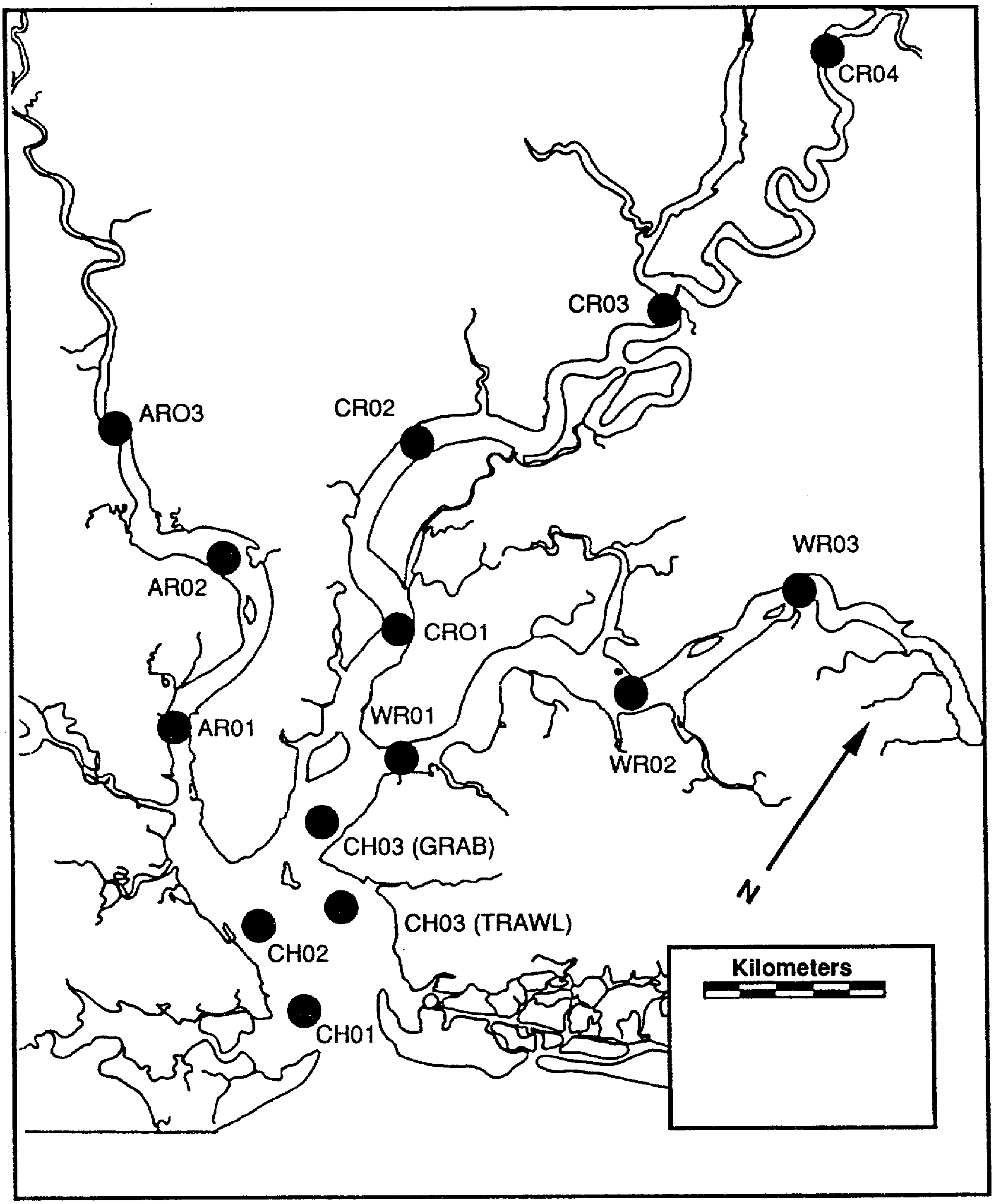

Figure III.3. Station locations for trawl and grab sampling.

Statistical analyses for differences in parameter means were conducted utilizing parametric and non-parametric analysis of variance (ANOVA) where appropriate. Statistical analyses were performed using Statistical Analysis Software (SAS) on a Data General minicomputer, and SYSTAT and BIOSTAT on an IBM-AT compatible computer. All statements of significant differences refer to an alpha=.05 level. Differences between stations for the 1988 annual mean dissolved oxygen, dissolved oxygen percent saturation, temperature, ortho-phosphate, nitrate, nitrite, ammonia and turbidity were determined using a Model I ANOVA (Sokal and Rohlf, 1981) after appropriate transformations were performed and if assumptions were met. Post hoc tests were applied to analyses indicating significant differences utilizing Tukey's HSD multiple comparisons test. Raw data values for temperature and dissolved oxygen were used in the ANOVA, while common log transformed values of ortho-phosphate and turbidity were utilized. The dissolved oxygen percent saturation was subjected to an arcsine transformation before statistical analyses and nitrate, nitrite and ammonia were transformed using an inverse hyperbolic sine function prior to ANOVA.

Comparisons of the post-rediversion data collected during this study with previously collected data (primarily through the Estuarine Survey Study) resulted in a high degree of heteroscedasticity which could not be reduced using data transformations. A Kruskal-Wallis two-way ANOVA was therefore utilized to determine if there were statistically significant differences in pre- and post-rediversion hydrographic parameters at comparable stations. Analyses which resulted in significant differences, and which had equal sample sizes, were subjected to post hoc tests utilizing the Sum of Squares Simultaneous Test Procedure (SS-STP).

RESULTS AND DISCUSSION

Summary statistics for all hydrographic data are provided in Appendix III.2. Characterizations and comparisons with other data sets were made using only the hydrographic sampling data which was standardized by tidal stage.

Freshwater Flow:

The daily mean flow of freshwater into the Cooper River through the Jeffries Dam averaged 121 m^3/s and ranged from 0 to 433 m^3/s during the period October, 1985 - December, 1988, while flow averaged 396 m^3/s, and ranged from 0 to 792 m^3/s during the period January, 1983 - September, 1985 (Figure III.4). Comparisons of monthly average flows during 3 pre-rediversion years (1982-1984) with those of 3 post-rediversion years (1986-1988) demonstrates distinct seasonal trends before rediversion which are lacking

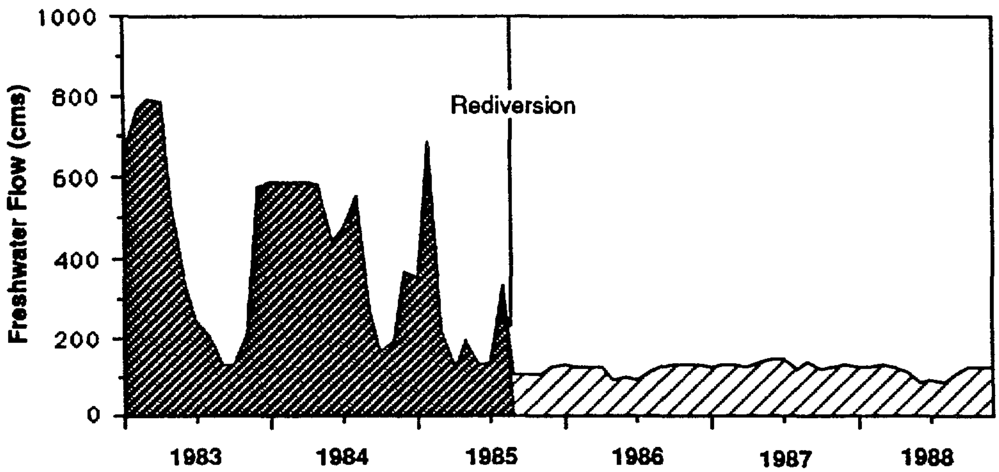

Figure III.4. Pre- and post-rediversion flow of freshwater into the Cooper River through the Jeffries Dam at Pinopolis.

in the post-rediversion years (Figure III.5). Flows were generally highest during the winter months before rediversion, decreasing in the spring and summer, with lowest flows occurring in the fall. Post-rediversion monthly and weekly mean flows were practically constant over the course of a year and seasonal trends have been negligible since rediversion. It should be noted, however, that mean daily flows often fluctuated as much as 216 cms diurnally.

Temperature:

Water temperature ranged from 3.5°C to 30.7°C and averaged 18.9°C during the period 1985-1988. Figure III.6 shows the seasonal and yearly changes in mean water temperature during the four-year study period. Monthly mean temperatures ranged from 6.2°C in January to 29.0°C in August during 1988 (Figure III.7). Bottom water temperatures were slightly lower than surface water temperatures during the period 1985-1988, and surface temperatures averaged 1.2°C higher than bottom temperatures. Prior to rediversion, water temperatures within Charleston Harbor averaged 19.8°C and ranged between 6.2 and 29.9°C throughout the year (USACOE, 1966; Mathews and Shealy, 1978, 1982; Kjerfve and Magill, 1990). Prior to rediversion, large diurnal variations (> 3°C) in temperature were not reported in the Charleston Harbor estuary, although differences

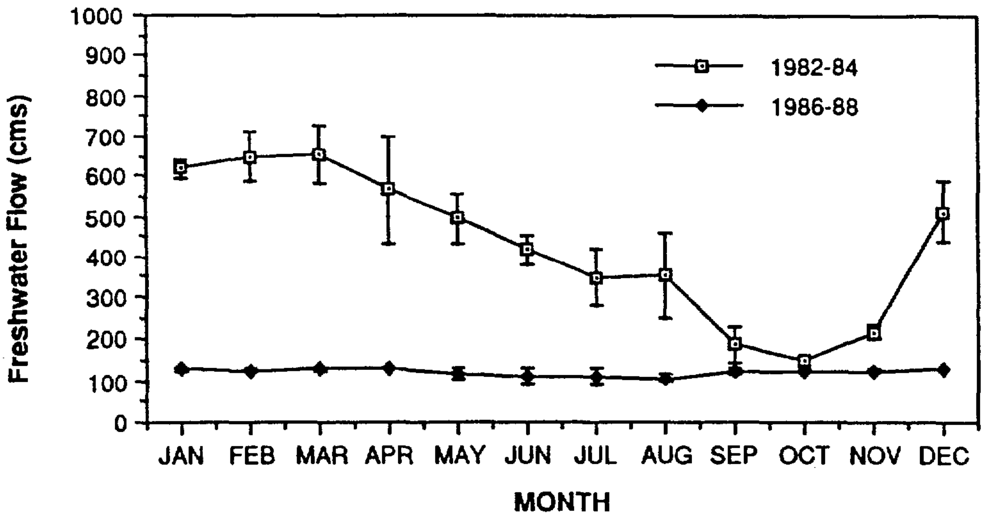

Figure III.5. Seasonal trends in freshwater flow into the Cooper River during a pre-rediversion period (1982-1984) and a post-rediversion period (1986-1988). Error bars are +/- 1 standard error of the mean.

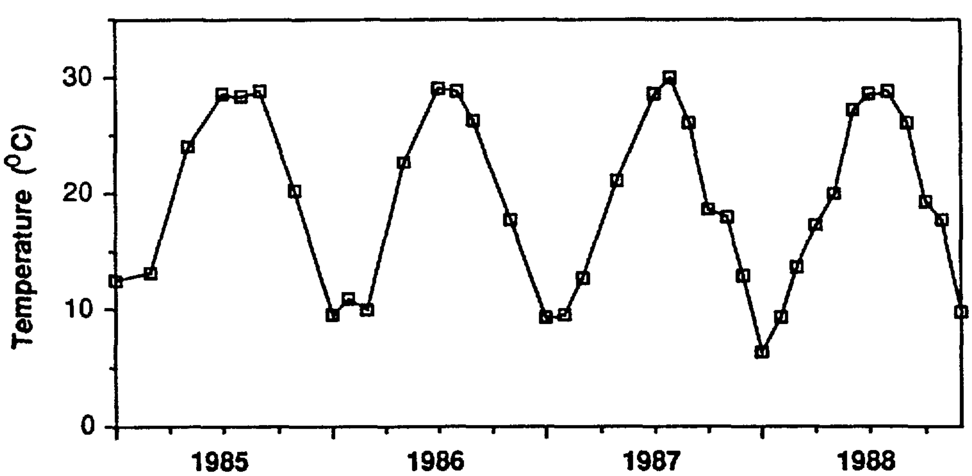

Figure III.6. Mean water temperature (surface and bottom combined) from all stations in the Charleston Harbor estuary during the period 1985-1988.

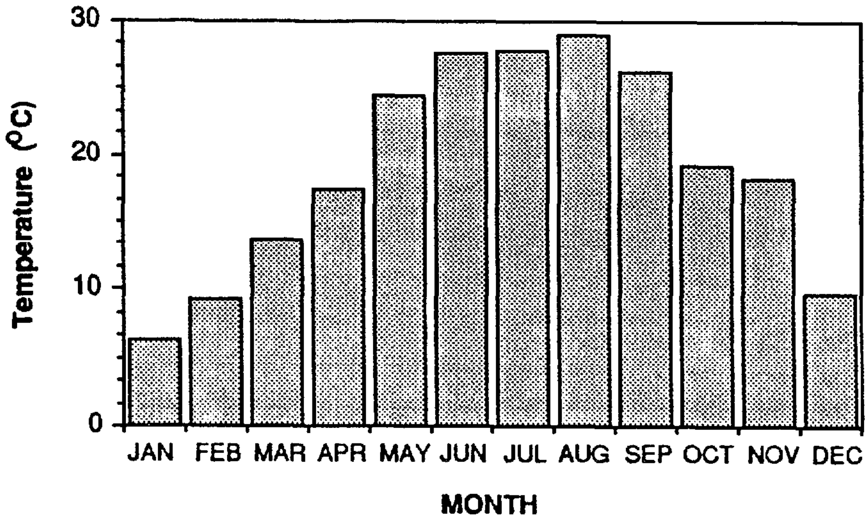

Figure III.7. Mean monthly water temperatures (all stations and depths combined) for the Charleston Harbor estuary during 1988. Standard error bars were negligible.

between surface and bottom temperatures ranged between 0.5 and 2.0°C (USACOE, 1966, 1972; Mathews and Shealy, 1978, 1982). According to Mathews and Shealy (1978, 1982), the average diurnal variation in water temperature during the period 1973 - 1978 was 1.5 °C, and the maximum difference ranged from 2.5°C at the surface to 2.7°C on the bottom. In addition, SCDHEC monitoring (Chestnut, 1989; Davis and Van Dolah, 1990) during the period 1970-1985 revealed a range of water temperatures from 1.5 to 35.0°C throughout the estuary.

Annual mean water temperatures exhibited no significant geographic trends in the estuary during the 1988 intensive sampling period (Figure III.8). Likewise, quarterly temperature values obtained during the grab sampling show similar geographic and seasonal trends in the estuary for the period 1985-1988 (Figure III.9).

Salinity:

During 1988, saline waters (< 0.5 ppt) extended approximately 45 km up the Cooper River, on average, and beyond the uppermost sampling sites in the Ashley River (37.0 km) and Wando River (34.4 km). The 1988 combined surface and bottom and high and low tide annual mean salinities by basin demonstrate distinct geographic trends within each

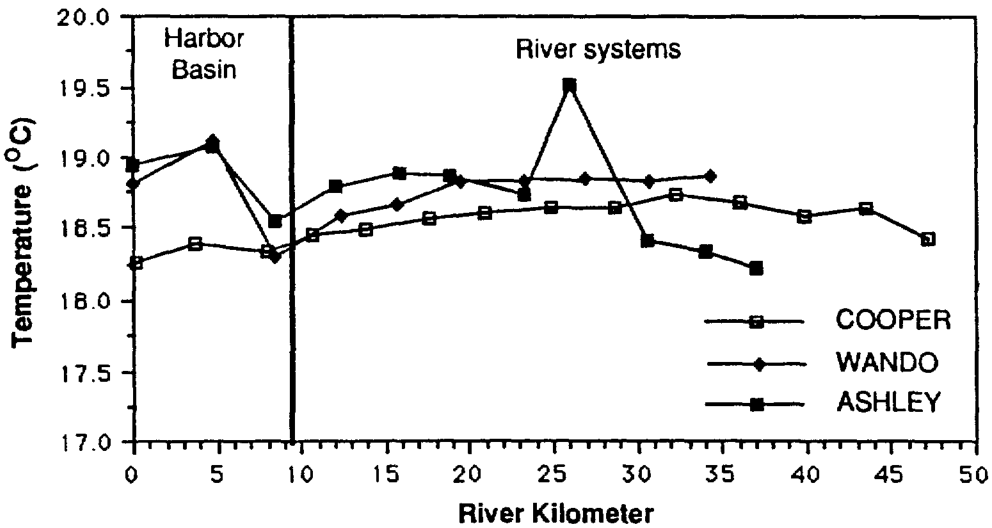

Figure III.8. Annual mean water temperatures for all four basins of the Charleston Harbor estuary during 1988. Values are mean of surface and bottom, and high and low tide samples. Standard error bars exceeded the graphs range and are not depicted due to their large size.

system (Figure III.10). The harbor basin exhibited a polyhaline salinity regime, while the Cooper River exhibited a salinity regime which ranged from polyhaline in its lower reaches to limnetic in its upper reaches. Salinity regimes in the Ashley River ranged from polyhaline to oligohaline (Figure III.10), while the Wando River exhibited a salinity regime which was almost entirely polyhaline. Mathews and Shealy (1978, 1982) reported that saline conditions extended approximately 29 km up the Cooper River from the mouth of the harbor prior to rediversion, and that a distinct salt wedge extended upstream approximately 15 km. Average salinities (combined surface and bottom) were reported to be 10.4 ppt at the mouth of the Cooper River, and 23.1 ppt at the mouth of the harbor during the period 1973-1978 (Mathews and Shealy, 1978, 1982), as compared with 23.0 ppt and 28.5 ppt in 1988.

The mean surface salinity in the lower harbor basin was approximately 26 ppt during the period 1986-1988 and declined gradually in the upstream direction on the Cooper River to station COM (RK 43.5) where it became limnetic (Figure III.11). Bottom salinities averaged approximately 32 ppt at the mouth of the harbor and also decreased gradually in

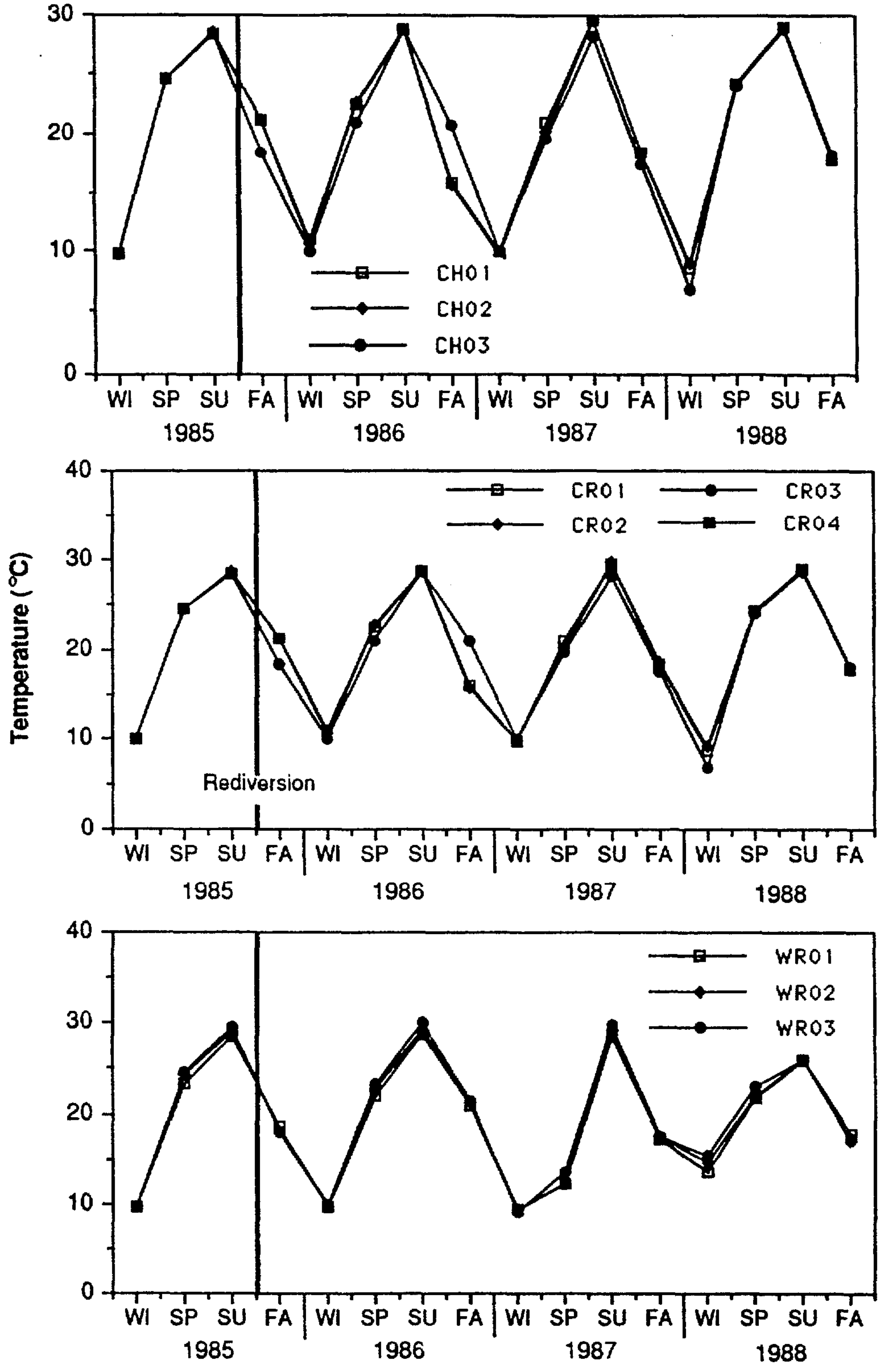

Figure III.9. Quarterly bottom water temperature values for the grab sampling sites in the harbor basin, and Cooper and Wando Rivers for the period of 1985-1988. The Ashley River was sampled only five times during this period, and the data are not presented here.

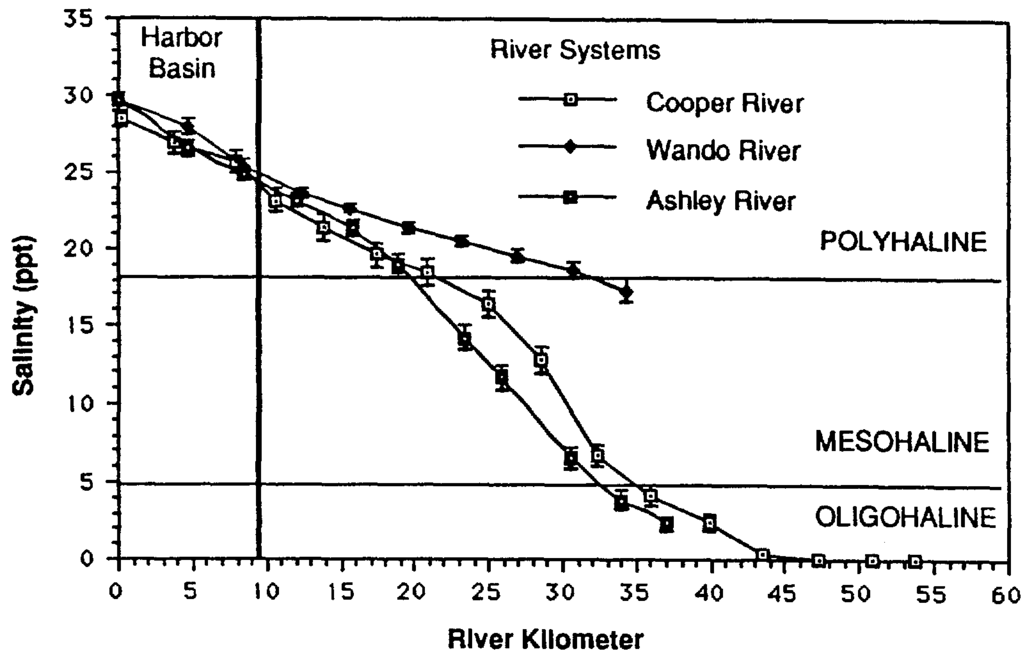

Figure III.10 Annual mean combined surface and bottom, and high and low salinities at all hydrographic stations in the Charlston Harbor estuary for 1988. Error bars are +/- 1 standard error of the mean.

the upstream direction on the Cooper River to station COH (RK 24.9), and then decreased more rapidly until the freshwater point at station COL (RK 39.8). In contrast, Mathews and Shealy (1978, 1982) reported that bottom salinities decreased from approximately 27 ppt at the mouth of the harbor to freshwater approximately 35 km upstream in the Cooper River prior to rediversion. The Cooper River appeared to be more stratified (surface to bottom) between stations COD (RK 10.6) and COI (RK 28.6) than other areas of the estuary. The Ashley River appears to be a more mixed, less stratified system than the Cooper River (Figure III.11) due to much lower freshwater flow. Gradual decreases in both surface and bottom salinity occurred in the upstream direction, and no dramatic decreases in salinity were observed as they were in the Cooper River. The Wando River exhibited a salinity regime similar to that of the Ashley River in that it also appears to be less stratified than the Cooper River, although the decrease in salinity is much more gradual than in the Ashley River. This, again, is due to much lower freshwater flows in the Wando River than the Cooper River, and somewhat lower flows than in the Ashley River.

The difference in mean surface and bottom (combined) salinities between high and low tides averaged 3.7 ppt throughout the estuary during 1988 (Figure III.12). Mean surface

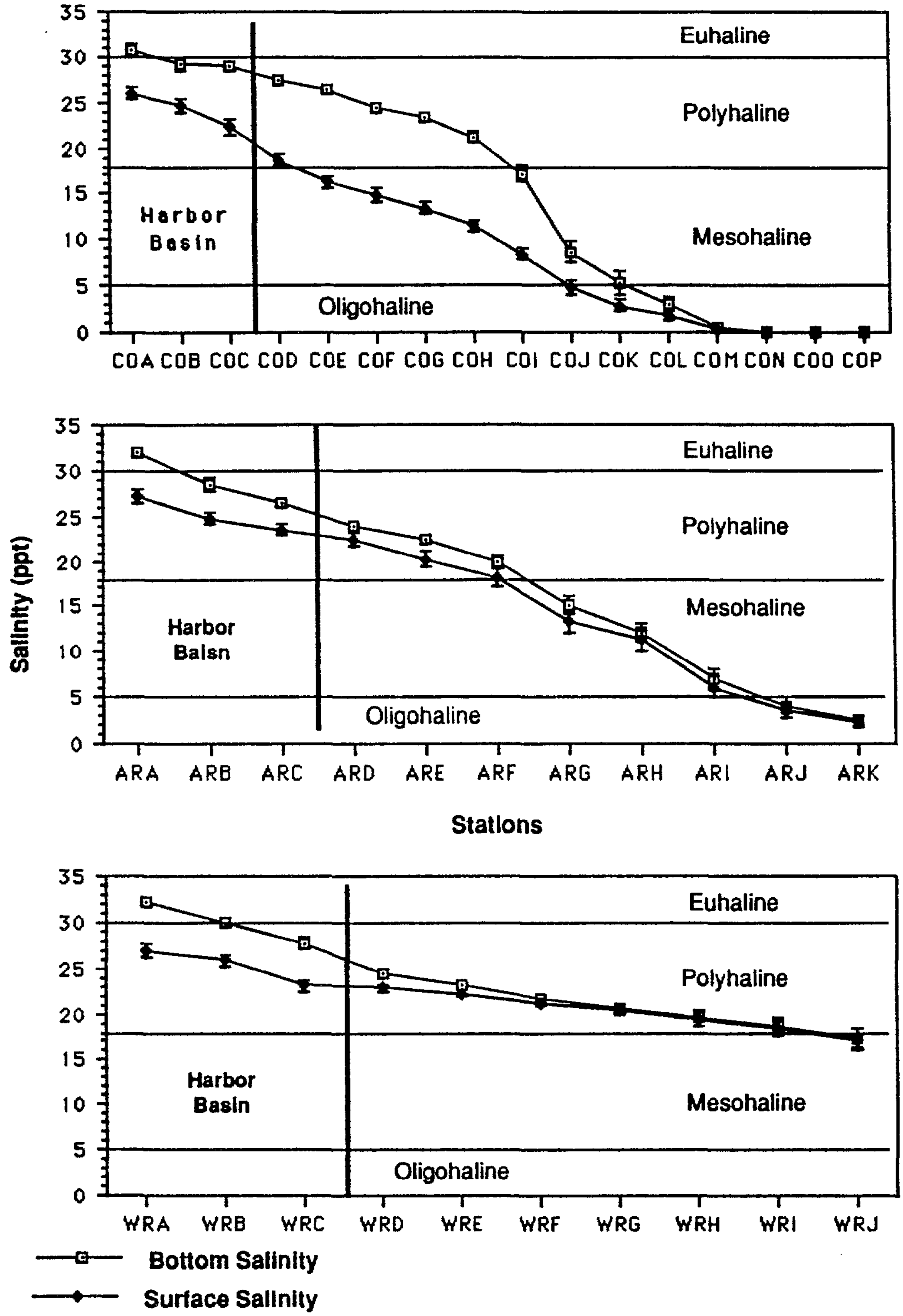

Figure III.11 Mean annual surface and bottom salinities at hydrographic stations in the Charleston Harbor estuary for 1988. Each value represents combined high and low tide samples, and the error bars are +/- 1 standard error of the mean.

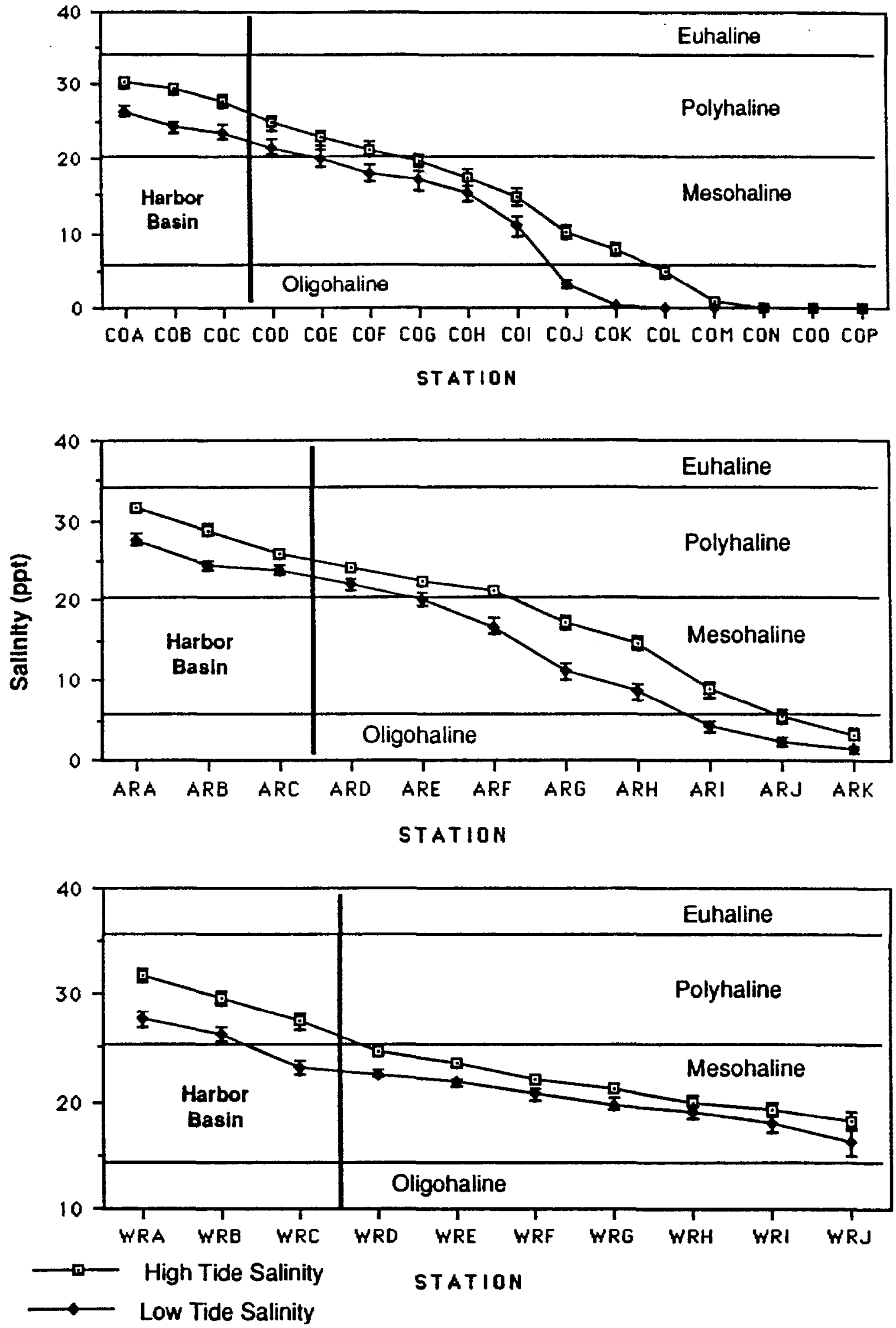

Figure III.12. Mean annual high and low tide salinities at hydrographic stations in the Charleston Harbor estuary for 1988. Each value represents combined surface and bottom salinity samples, and the error bars are +/- 1 standard error of the mean.

and bottom salinities at high tide averaged 3.2 ppt higher than low tide salinities in the harbor basin. The lower Cooper River stations (COD-COH) exhibited smaller changes in salinity between high and low tides, on the order of 1-2 ppt (Figure III.12), while stations further upriver (COI-COL) exhibited changes of 5 to 8 ppt. Stations COM through COP exhibited salinity regimes which were almost entirely freshwater. Stations in the lower Ashley River (ARD and ARE) exhibited tidal changes in salinity of 1-3 ppt, as did the uppermost stations (ARJ and ARK). Stations ARG and ARH exhibited changes of approximately 7 ppt, while changes in salinity at stations ARF and ARI were approximately 5 ppt. Changes in salinities between high and low tides averaged approximately 2 ppt at all Wando River stations.

A comparison of mean isohaline surfaces in the Cooper River during the 1988 intensive sampling with data obtained prior to rediversion by Mathews and Shealy (1982), the SCEHEC (Chestnut, 1989), and the USGS (unpublished data available through USEPA's STORET system) demonstrates that the surface freshwater isohaline (< 0.5 ppt) was located approximately 6 km further upstream, while the bottom freshwater isohaline was located approximately 2.5 km further upstream during the 1988 period (Figure III.13). The isohalines in the Cooper River have become more separated when compared with pre-rediversion data, and the slopes of the isohalines are less pronounced when comparing complementary (by salinity) regions of the river.

A four-year comparison of low tide, bottom salinities at selected stations in the harbor basin and Cooper and Wando Rivers demonstrates the seasonal, long-term variability within the estuary (Figure III.14). Although the winter, 1985 samples were not all collected at low tide, it appears that salinities in the harbor basin, the Wando River, and the lower Cooper River were lower during this period than post-rediversion winter salinities. This may be due, in part, to drought conditions which occurred during the winter and summer of 1985. In 1985 salinities during the summer were also lower than salinities during the post-rediversion summer periods at many stations, although the spring salinities were not. This may be due to drought conditions being temporarily relieved when heavy rains occurred during late spring in 1985.

Seasonal trends in salinity were not observed in the Charleston Harbor estuary during post-rediversion sampling, with the exception of slightly depressed salinities in the upper Ashley and Wando Rivers during both high and low tides in September, 1988 (Figure III.15). The Cooper River exhibited no seasonal trends in salinity, although distinct seasonal trends in salinity were reported prior to rediversion (Kjerfve and Magill, 1990). This lack of seasonal trends in salinity in the Cooper River and the Charleston Harbor basin is attributable to the reduced flow, and more stable flows through the Jeffries Dam

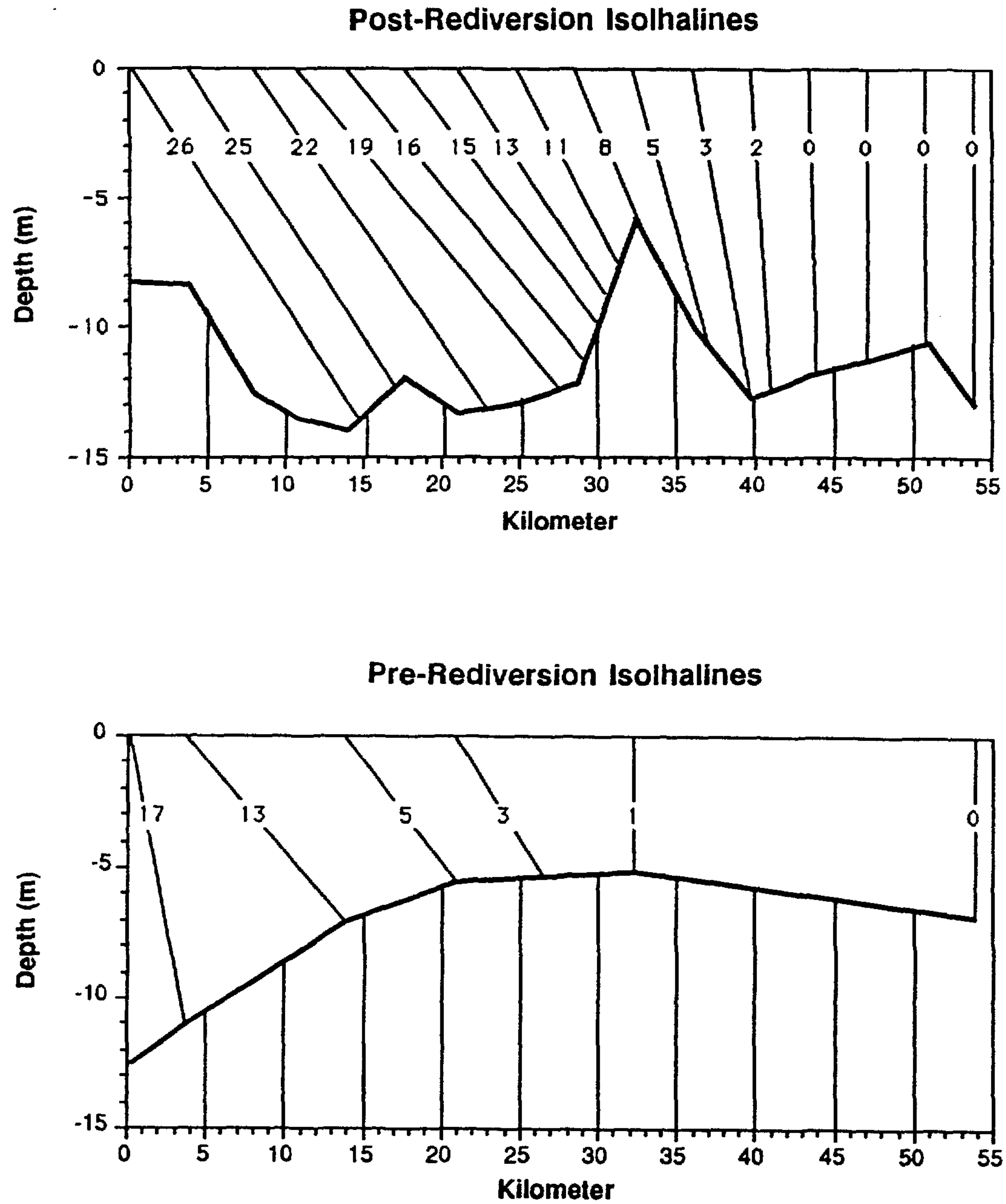

Figure III.13. Mean high and low tide isohaline surfaces from the mouth of the harbor to the upper reaches of the Cooper River under post-rediversion conditions (top) and pre-rediversion conditions (bottom). Post-rediversion isohalines are the result of averaging high and low tide salinities from 12 monthly samplings in 1988 at 16 stations. The pre-rediversion isohalines are adapted from several data sources (see text). Differences in the bottom topography of the two graphs are due to accurate reproduction of the pre-rediversion graph from Mathews and Shealy (1982), and do not represent any changes in the basin.

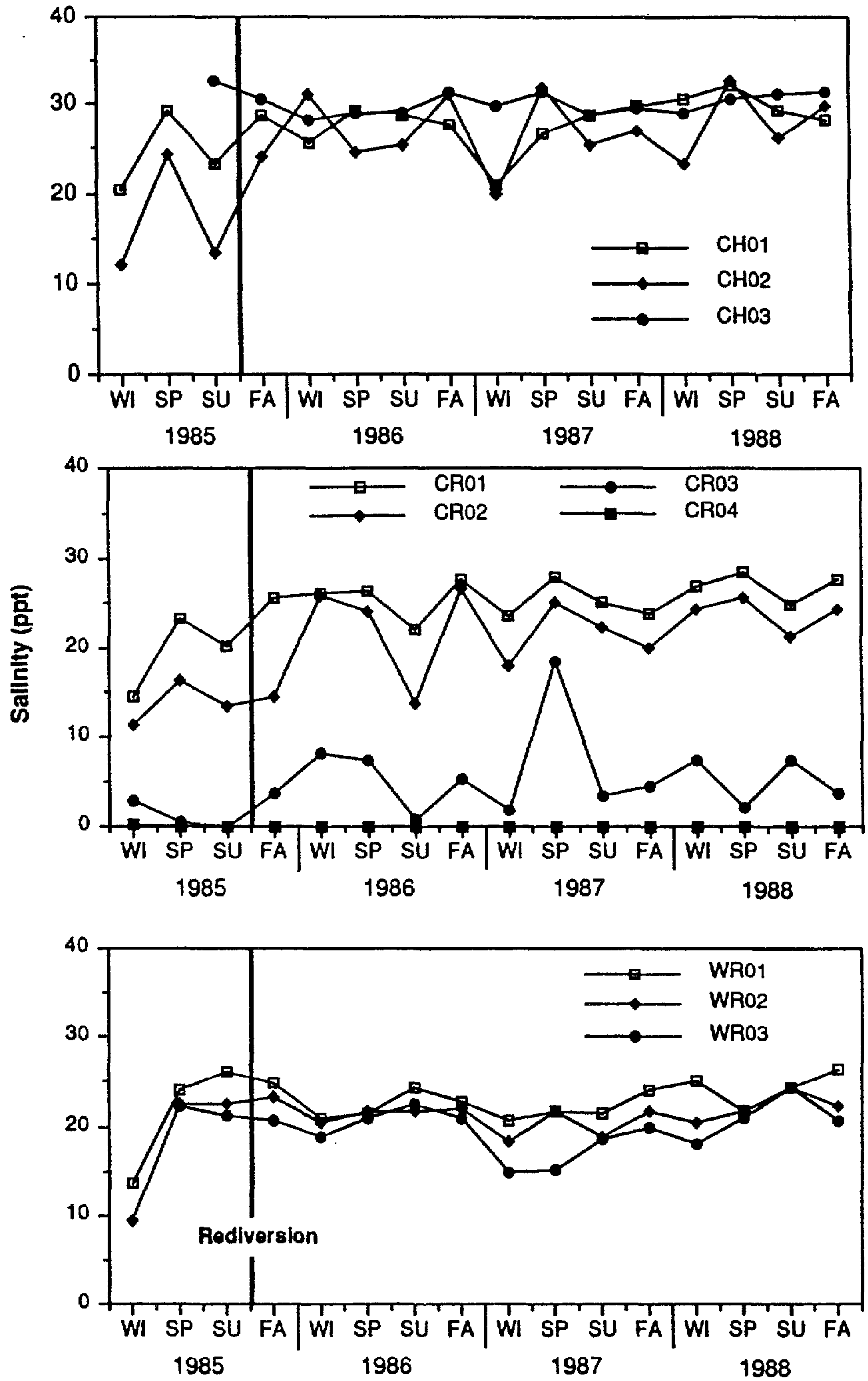

Figure III.14. Quarterly bottom salinity from grab sampling sites in the harbor basin and Cooper and Wando Rivers during the period 1985-1988. The Ashley River grab sites were sampled only five times during this period.

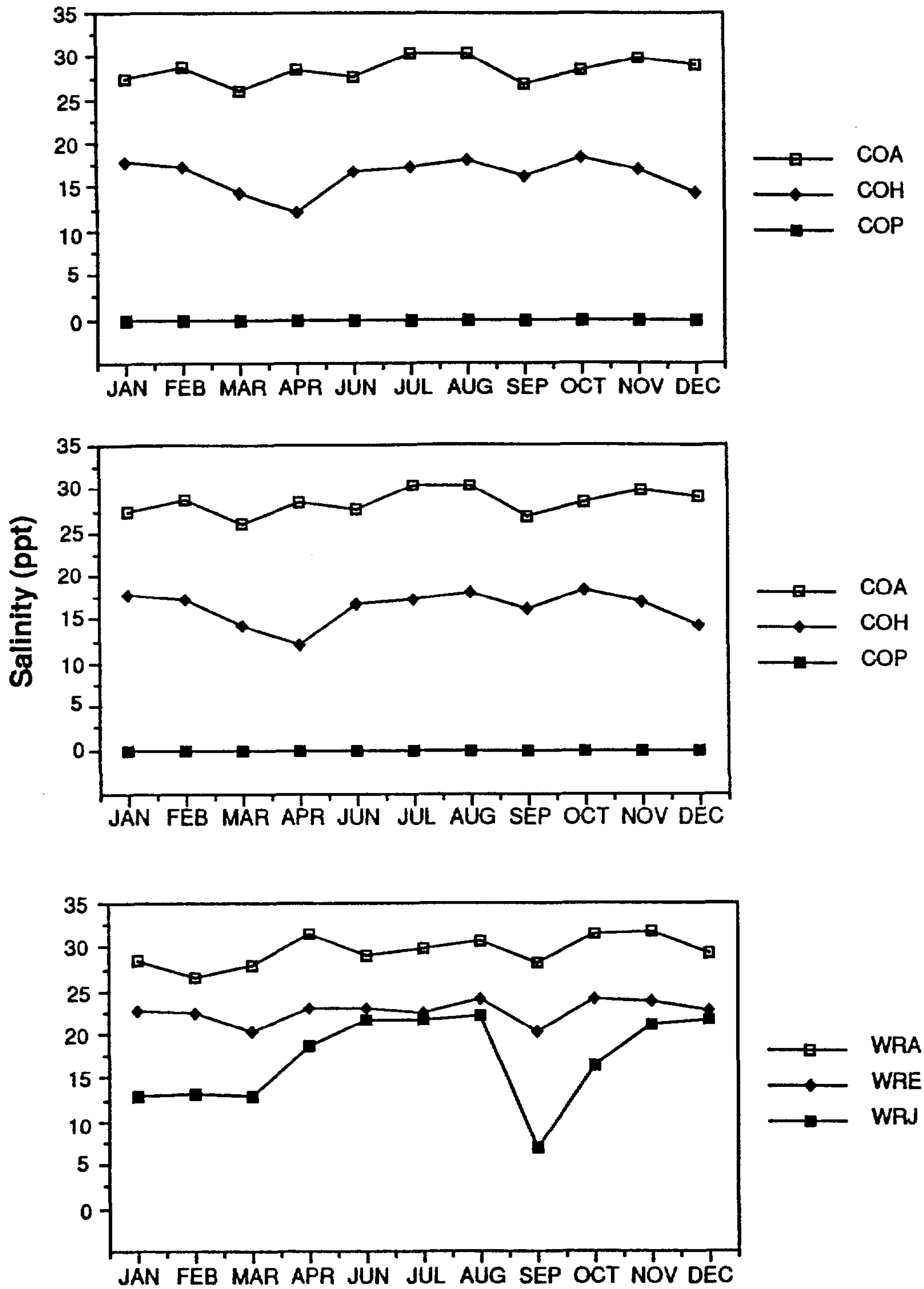

Figure III.15. Seasonal variation in mean surface and bottom, and high and low tide salinities at selected stations in the harbor basin, and Cooper, Ashley and Wando Rivers during 1988. May salinity data was omitted due to instrument malfunction. Error bars are not depicted due to their large size.

at Pinopolis South Carolina. Analyses for correlation between one, two, three and four-day mean flows with salinity in the upper Cooper River yielded negative results even though consecutive mean daily flow rates often fluctuated by as much as 8,000 cfs. Similar results were obtained in the James River Estuary by Haas (1977), where he demonstrated that salinity regimes were completely independent of freshwater flow.

Prior to rediversion, salinity regimes in Charleston Harbor were predominantly controlled by freshwater flow, and exhibited distinct seasonal trends (FWPCA, 1966; USACOE, 1966; Kjerfve, 1988). At high river discharges the estuary was strongly stratified and salinity distribution was dependent on the tidal stage and amplitude. At freshwater flows less than 280 m^3/s, the estuary was less vertically stratified (FWPCA, 1966; USACOE, 1966, 1972; Kjerfve and Magill, 1990). The rate at which salinity moved upriver was influenced by the tidal range as well as by the prevailing downstream flow (FWPCA, 1966; SCWRC, 1979). Differences in tidal amplitude had a pronounced effect on salinity distribution (Van Nieuwenhuise, 1978).

One interesting anomaly in the pre-rediversion circulation in the Charleston Harbor estuary, which was not found during the post-rediversion period, occurred in the Wando River. High and low slack tides occurred in the southerly portion of the Wando River approximately 40 minutes before they occurred in the Cooper River, and at low slack tide the flow of water from the Cooper River often moved upstream into the Wando River (USACOE, 1966). Similarly, at high slack tide, water flowed up the Cooper River from the Wando River. Under certain conditions, the salinity 13 km up the Wando River was reported to be higher than that encountered at its mouth (USACOE, 1966).

Dissolved Oxygen:

Bottom dissolved oxygen (D.O.) concentrations in the Charleston Harbor estuary ranged from 2.56 to 12.81 mg/l and averaged 7.09 mg/l during the 1986-1988 post-rediversion survey period, and exhibited both geographic and seasonal trends. SCDHEC monitoring during the period 1970-1985 revealed that D.O. concentrations ranged from 0.0 to 17.1 mg/l, with an average of 7.5 mg/l for the entire estuary (Davis and Van Dolah, 1990). Little (1974) stated that D.O. concentrations between 4.9 and 9.4 mg/l had been reported in bottom waters of the estuary. Differences in D.O. concentrations between low and high tide were negligible during the 1986-1988 survey period. The 1988 mean D.O. concentration appeared to be somewhat lower in the upper Ashley River than other areas of the estuary, and higher in the upper Cooper River (Figure III.16), although the differences were not statistically significant.

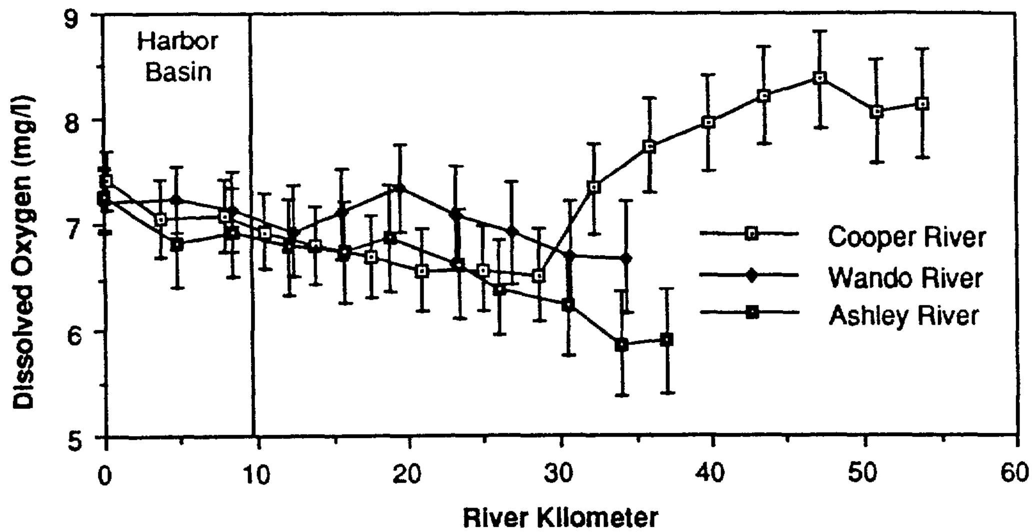

Figure III.16. Annual mean bottom dissolved oxygen at hydrographic sampling stations in the Charleston Harbor estuary for 1988. Values are mean of 12 high and 12 low tide samples obtained monthly during 1988. Error bars are +/- 1 standard error of the mean.

Analysis of D.O. concentrations did not result in significant differences between stations in the estuary except between the uppermost Ashley River stations (ARJ and ARK) and the Cooper River station CON. Dissolved oxygen concentrations are affected by both salinity and temperature, and these parameters changed in the upstream direction during individual sampling periods. Salinity changed dramatically in the upstream direction, while temperatures generally were higher in the upstream direction due to the fact that hydrographic sampling was always initiated in the morning at the mouth of the harbor, and the upper stations were sampled later in the day. The temperature and salinity compensated D.O. percent saturations were, therefore, utilized to further examine the D.O. regimes within the estuary.

Results of the ANOVA on arcsine-transformed, 1988 D.O. percent saturations demonstrated a significantly lower mean D.O. percent saturation at station COI compared with the lower harbor stations COA, WRA, WRB and ARA. In addition, the uppermost Wando River stations (WRI and WRJ) exhibited significantly lower mean D.O. percent saturations than the same lower harbor stations. The two uppermost Ashley River stations (ARJ and ARK) had significantly lower mean D.O. percent saturations than all other areas of the estuary, with the exceptions of adjacent Ashley River stations and station COI on the Cooper River. According to the FWPCA (1966), low D.O. concentrations (less than 3 mg/l) were commonly reported from the upper Ashley River during the 1950's and 1960's, and it appears that this trend is continuing.

The 1988 mean D.O. concentration and percent saturation gradually decreased in the upstream direction in both the Ashley and Wando Rivers (Figure III.17). Bottom D.O. concentrations in the harbor basin averaged 7.3 mg/l and decreased to 6.8 mg/l in the upper Wando River and 6.1 mg/l in the upper Ashley River. The Wando River also exhibited a slight increase in both D.O. concentration and percent saturation at the mid stations of the sampling transect. The mean percent saturation of bottom D.O. was approximately 90% in the harbor basin and decreased to approximately 78% in the upper Wando River. The bottom D.O. percent saturation in the Ashley River steadily decreased to a low of approximately 62% in the upper Ashley River, the lowest in the estuary. The reduced D.O. concentrations and percent saturation in the upper Ashley River are most likely a result of high nutrient loading in the system. High concentrations of nutrients and organic material are dumped into the upper Ashley River through municipal sewage facilities and urban and rural runoff.

In contrast with the Ashley River, the D.O. concentration in the Cooper River declined gradually to approximately 6.4 mg/l at station COI (RK 28.6), and then rapidly increased in the upstream direction to approximately 8.0 mg/l at station COM (RK 43.5). The bottom D.O. percent saturation exhibited a similar, although more gradual, trend, decreasing to approximately 73% at station COI (RK 28.6), and then increasing to approximately 85% in the upstream direction. The USACOE (1966) reported percent saturation of D.O. in bottom waters of 52% in the upper harbor, and 77% in bottom waters of the lower harbor. Mathews and Shealy (1978) reported mean D.O. percent saturations of 80% near the mouth of the Cooper River, and 90 to 95% at the mouth of the harbor.

A comparison of 1988 annual average bottom D.O. concentrations and percent saturation with 1973 - 1978 Estuarine Survey Data (Mathews *et al.*, 1981; Figure III.18) demonstrated geographic trends in the harbor basin and Cooper River similar to those found during post-rediversion sampling (Figure III.17). A Kruskal-Wallis ANOVA found no significant differences between the pre- and post-rediversion mean D.O. or D.O. percent saturations. Only one station was sampled in each of the Ashley and Wando Rivers during the Estuarine Survey Study, but equivalent stations sampled in 1988 exhibited similar values for both D.O. concentrations and percent saturation. Likewise, a comparison of post-rediversion D.O. values (1986-1988) from selected stations within the harbor with pre-

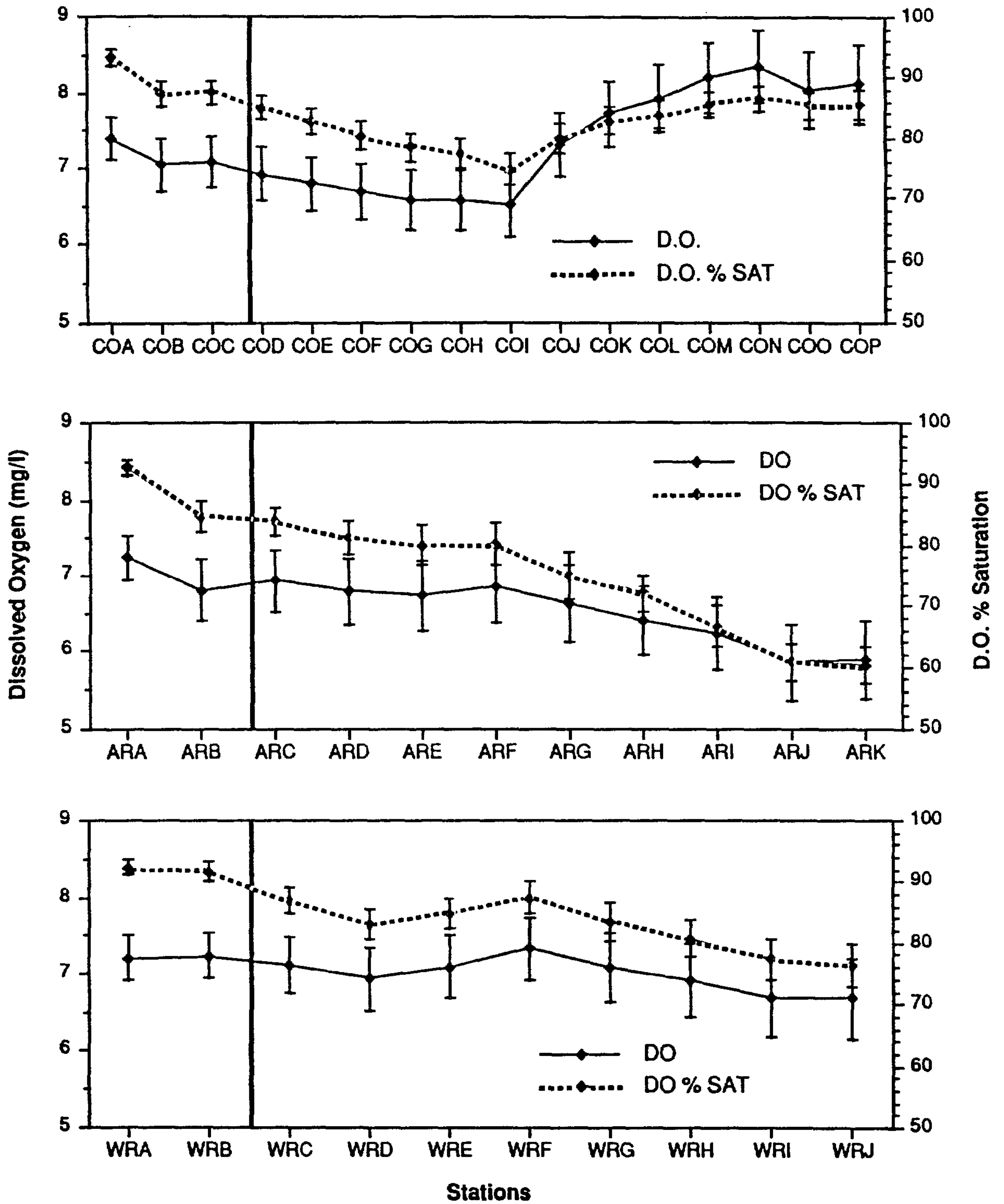

Figure III.17. Annual mean bottom dissolved oxygen and percent saturation at hydrographic sampling stations for 1988. Error bars are +/- 1 standard error of the mean.

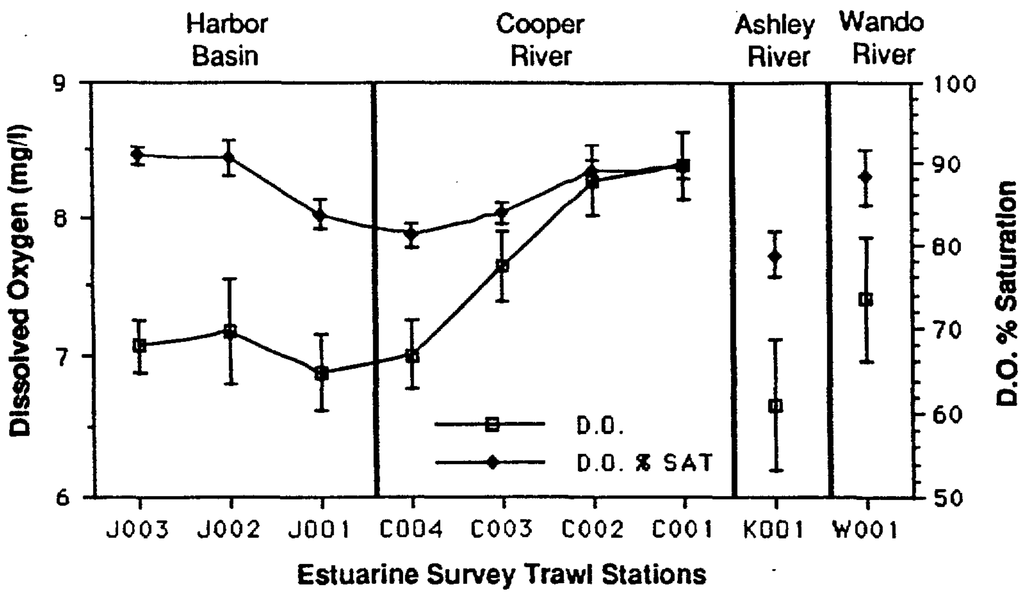

Figure III.18. Mean bottom dissolved oxygen and percent saturation reported by the Estuarine Survey Study during the period 1973-1978 (Mathews and Shealy 1978, 1981, 1982). Error bars are +/- 1 standard error of the mean.

rediversion D.O. values obtained by SCDHEC (1970-1984) also demonstrated no significant differences at most stations in the estuary (Table III.1). One exception to this occurred at station ARF where the post-rediversion mean D.O. concentration was significantly higher during the post-rediversion period. These comparisons suggest that D.O. concentrations have not been reduced in the Cooper River or lower harbor basin as previously suspected might happen (USACOE, 1966), and indicate that post-rediversion D.O. regimes in the estuary are similar to pre-rediversion conditions.

The 1988 seasonal trends in the bottom D.O. concentration and percent saturation were similar in all 4 systems, and generally reflect the expected effects of temperature and biological respiration (Figure III.19). Dissolved oxygen concentrations averaged approximately 10 mg/l in January and gradually declined through May. The D.O. concentration remained relatively constant through July, decreased again in late summer, and then increased through the fall into winter. The lowest average D.O. concentrations for each basin occurred in the August-September period, and were approximately 5 mg/l in the Wando River, 4 mg/l in the Ashley River, 5 mg/l in the Cooper River, and 6 mg/l

Table III.1 A comparison of pre-rediversion bottom dissolved oxygen concentrations with post-rediversion concentrations at comparable stations. Pre-rediversion values were obtained through the SCDHEC monitoring program, and the number of years each station was sampled is indicated. Dissolved oxygen values are in mg/l. Post-rediversion concentrations are the mean of high and low tide hydrographic samples. Error terms are +/- 1 standard error of the mean.

	Pre-rediversion			Post-rediversion		
Station	Years	D.O.	S.E.	D.O.	S.E.	Station
MD-048	(72-84)	7.36	0.17	7.27	0.94	COA
MD-052	(70-84)	6.89	0.25	6.95	1.17	ARD
MD-135	(70-84)	7.13	0.26	5.34	0.87	ARF
MD-047	(70-84)	7.10	0.21	7.14	0.98	COC
MD-045	(74-84)	6.81	0.18	7.07	1.17	COE
MD-152	(72-84)	7.19	0.21	7.43	1.04	COJ
MD-502	(79-82)	7.37	0.19	6.63	1.24	WRE

in the harbor basin. The bottom D.O. percent saturation exhibited similar trends, with the January values averaging approximately 95% in all systems, and declining to approximately 65% in the Wando River, 50% in the Ashley River, 65% in the Cooper River, and 80% in the harbor basin during the August-September period. Previous investigations also reported that D.O. concentrations were generally higher in the colder months than in the summer months (FWPCA, 1966; USACOE, 1966; Little, 1974).

Results of the extensive four-year (1985-1988) quarterly trawl sampling demonstrate seasonal trends in bottom D.O. concentrations similar to those found during the intensive 1988 sampling period (Figure III.20). They also exhibit no differences between the harbor basin, and Cooper and Wando Rivers, nor between pre- and post-rediversion periods. In some instances, stations farther upstream appear to exhibit slightly lower values which supports earlier conclusions regarding the geographic distribution of D.O. within the estuary. It should be pointed out, however, that these trends may be due to a salinity effect, the time of day each station was sampled, or both. Sampling on any given day started in the

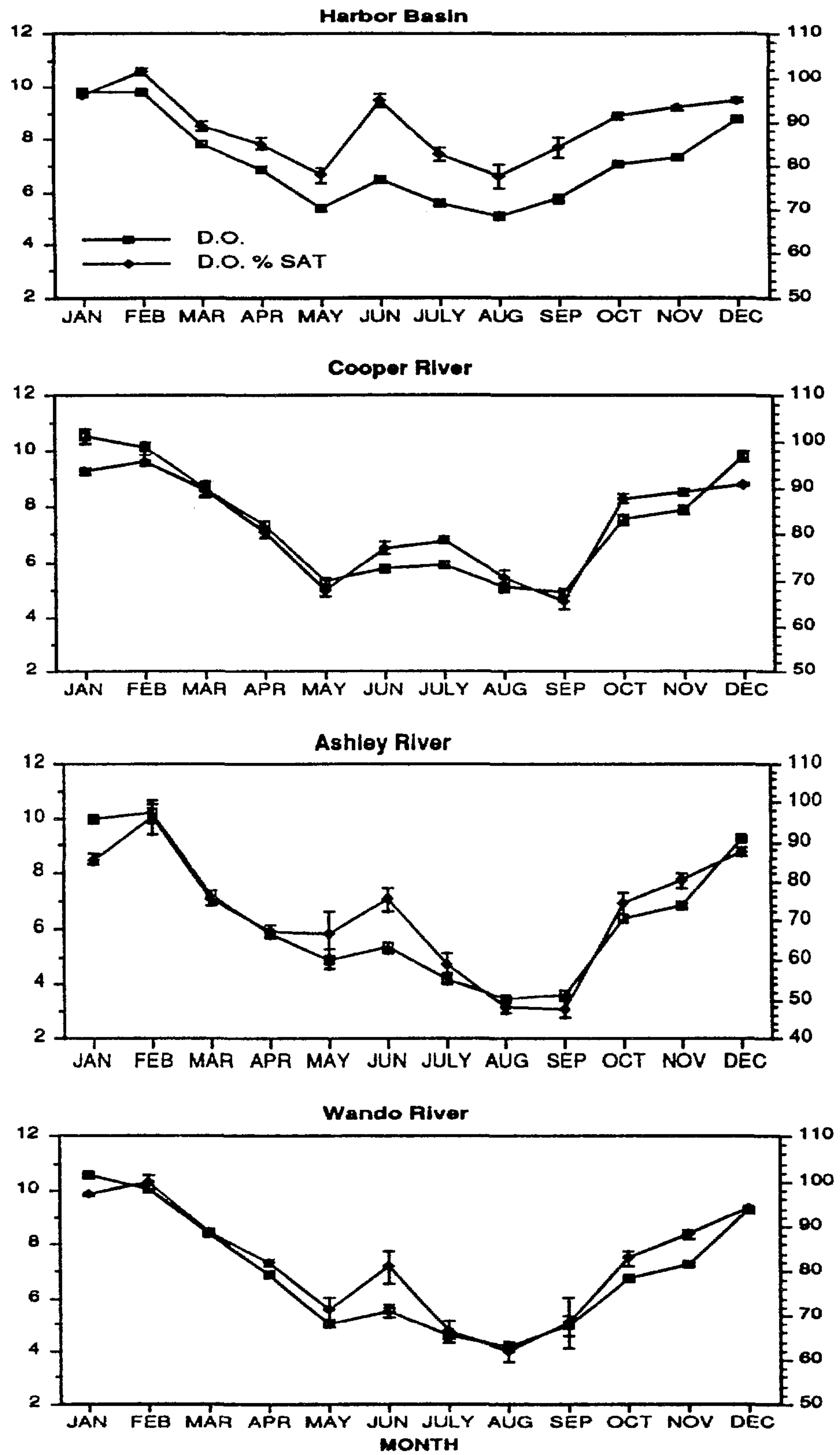

Figure III.19. Seasonal fluctuations in bottom dissolved oxygen and percent saturation during 1988 in the harbor basin and Cooper, Wando and Ashley Rivers. Error bars are +/- 1 standard error.

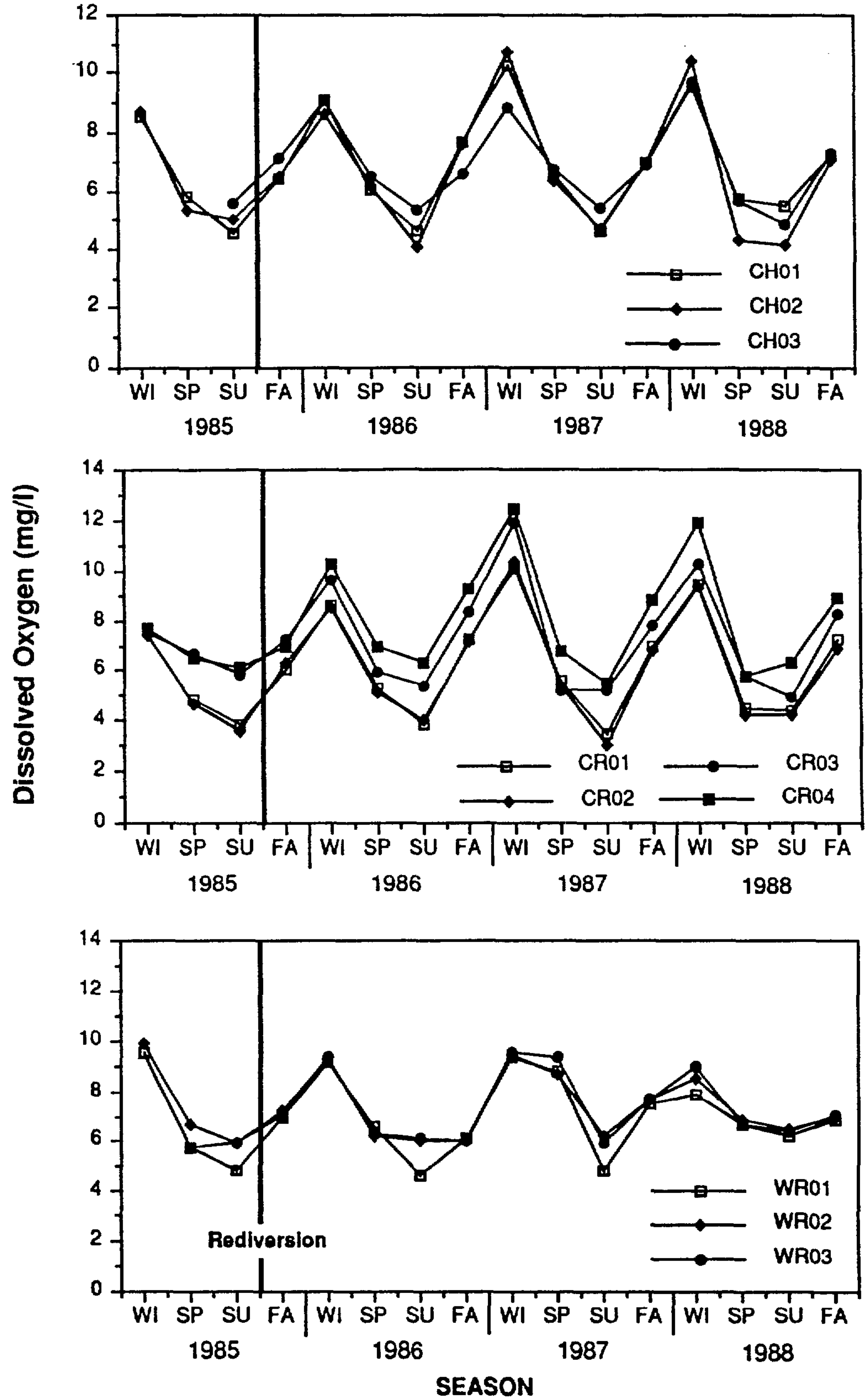

Figure III.20. Bottom dissolved oxygen concentrations found at quarterly trawl sampling stations in the harbor basin and Cooper and Wando Rivers during the period 1985-1988.

lower harbor and proceeded in an upstream direction, resulting in upper stations always being sampled later in the day than lower stations.

Dissolved oxygen levels in the Charleston Harbor estuary are influenced by many factors, including temperature, photosynthesis, respiration and mixing. Prior to rediversion, with high river flow and strong stratification, mixing between the surface and bottom layers was restricted, and the major source of D.O. for the bottom layer was offshore, oceanic waters (FWPCA, 1966). Consequently, the concentration of D.O. in the bottom layer was dependent on factors affecting bottom flow. Also, at high river flow, the D.O. percent saturation was reported to be fairly constant throughout the estuary (FWPCA, 1966; USACOE, 1966). At low river flow, surface aeration was reported to be the major source of D.O. throughout the estuary, and the D.O. concentration in the estuary was generally lower during low river flow, and dropped markedly in the upstream direction (FWPCA, 1966). Rediversion resulted in a major decrease in river flow in the Cooper River and the results of the post-rediversion hydrographic sampling support the earlier reports.

Turbidity:

Turbidity values ranged from 1.3 to 84.0 NTU during the period 1986 - 1988, and exhibited geographic trends (Figure III.21), and trends with depth, but no distinct seasonal trends (Figure III.22). Surface turbidity averaged 5.3 NTU and ranged from 1.3 to 24.0 NTU, while bottom turbidities averaged 11.0 NTU and ranged from 2.0 to 84.0 NTU. Average turbidities were highest in the Ashley River (12.8 NTU) and harbor basin (10.2 NTU) and lower in the Cooper (6.9 NTU) and Wando (6.1 NTU) Rivers. Turbidity values in the Harbor basin ranged from 1.6 to 84.0 NTU, Cooper River values ranged from 1.3 to 65.0 NTU, and Wando River values ranged from 1.8 to 34.0 NTU. Ashley River values, on the other hand, ranged from 1.5 to 36.0 NTU. Statistical analyses revealed that the upper Ashley River stations ARH and ARJ exhibited mean turbidities which were significantly higher than all other stations in the estuary. Mean turbidities at other stations in the estuary were not significantly different from one-another.

A comparison of turbidities obtained by the SCDHEC prior to rediversion with the post-rediversion turbidities revealed similar values at comparable stations with two exceptions (Table III.2). The post-rediversion mean turbidity value for station ARD was significantly higher than the pre-rediversion mean turbidity, while the post-rediversion value for station WRE was significantly lower than the pre-rediversion value. Likewise, data collected during the Estuarine Survey Study indicated no significant differences between pre- and post-rediversion mean turbidity values with the exception of the mouth of the

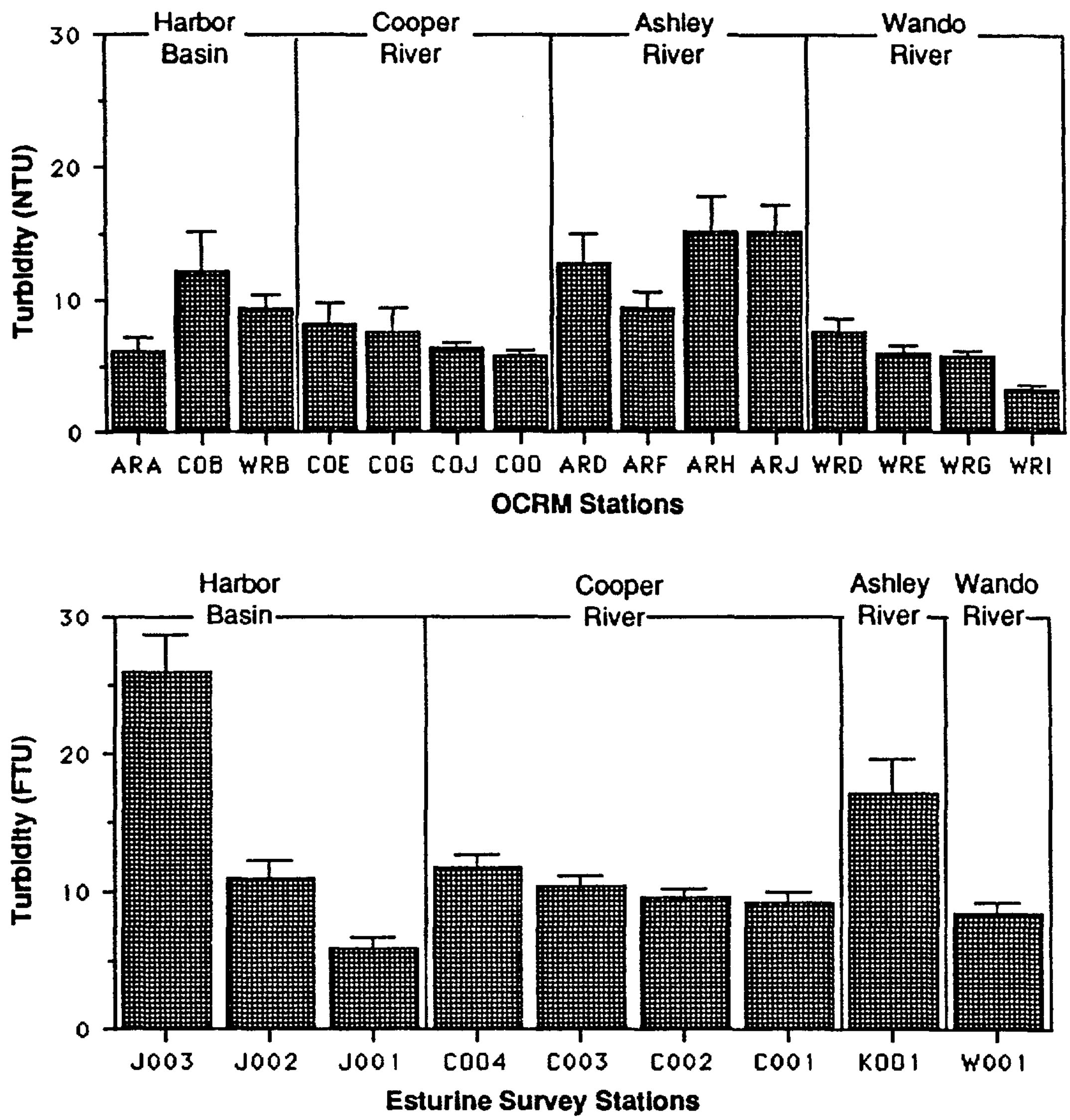

Figure III.21. Mean turbidity values for the period 1986-1988 at hydrographic sampling station and mean turbidity values for the period 1973-1978 at Estuarine Survey trawl stations in the Charleston Harbor estuary. Error bars are +/- 1 standard error of the mean.

Harbor, where the average turbidity value was 2 to 3 times higher during the 1973 - 1978 period than it was in the 1988 period (Figure III.21). The Estuarine Survey Study found a mean turbidity of 11.1 FTU for the entire estuary during the period 1973-1978, and the

Table III.2 A comparison of mean, surface and bottom combined pre-rediversion turbidities with post-rediversion turbidities at comparable stations. Pre-rediversion values were obtained through the SCDHEC monitoring program, and the number of years each station was sampled is indicated. Pre-rediversion turbidity values are reported as formazine turbidity units (FTU), while post-rediversion turbidity values are reported as nephelometric turbidity units (NTU). Error terms are +/- 1 standard error of the mean.

Station	Years	Pre-rediversion FTU	S.E.	Post-rediversion NTU	S.E.	Station
MD-052	(70-84)	9.25	0.74	13.51	0.71	ARD
MD-135	(70-84)	11.43	1.39	10.05	0.49	ARF
MD-045	(74-84)	8.80	1.21	9.10	0.53	COE
MD-152	(72-84)	7.57	0.74	7.15	0.30	COJ
MD-502	(79-82)	9.07	1.04	6.20	0.92	WRE

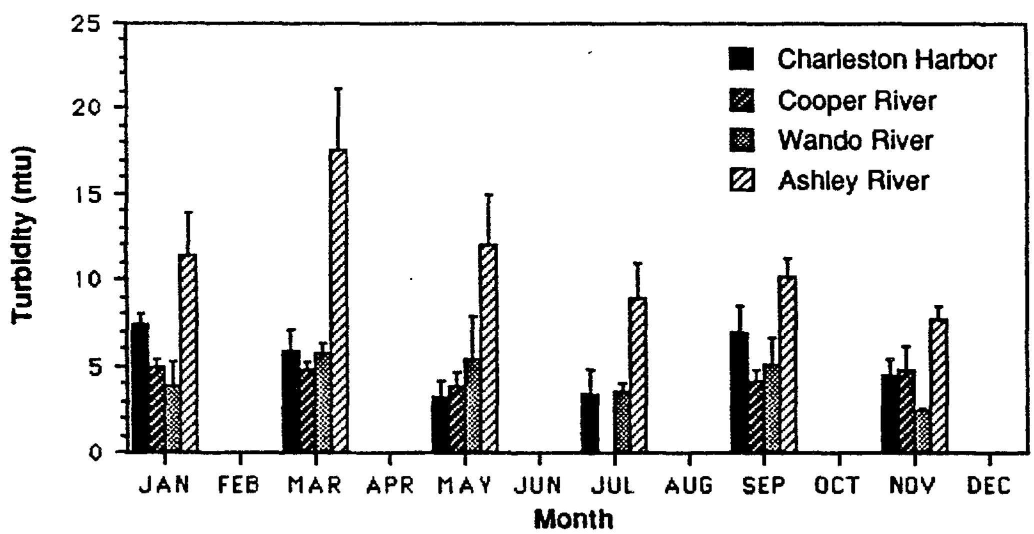

Figure III.22. Seasonal variability in turbidity from the harbor basin and Cooper, Wando and Ashley Rivers during 1988. Values are the mean of surface and bottom values from all stations in each basin. Error bars are +/- 1 standard error.

data also demonstrated that higher turbidities occurred in bottom waters than in surface waters, with a mean bottom turbidity of 13.5 FTU and a mean surface turbidity of 8.4 FTU.

Nutrients:

Distinct geographic trends and less pronounced seasonal trends were observed for nutrient concentrations in the Charleston Harbor estuary during the 1988 intensive sampling period. Annual averages for stations in the harbor basin, Cooper River, Wando River and lower Ashley River were similar for all four nutrients, while mid- and upper-Ashley River stations exhibited significantly higher concentrations of nitrates and phosphates than other areas of the estuary. Mean phosphate values at all Ashley River stations were significantly higher than all other areas of the estuary. In addition, concentrations of nitrites and ammonia at upper Ashley River stations were significantly higher than concentrations at many other stations throughout the estuary (Figure III.23).

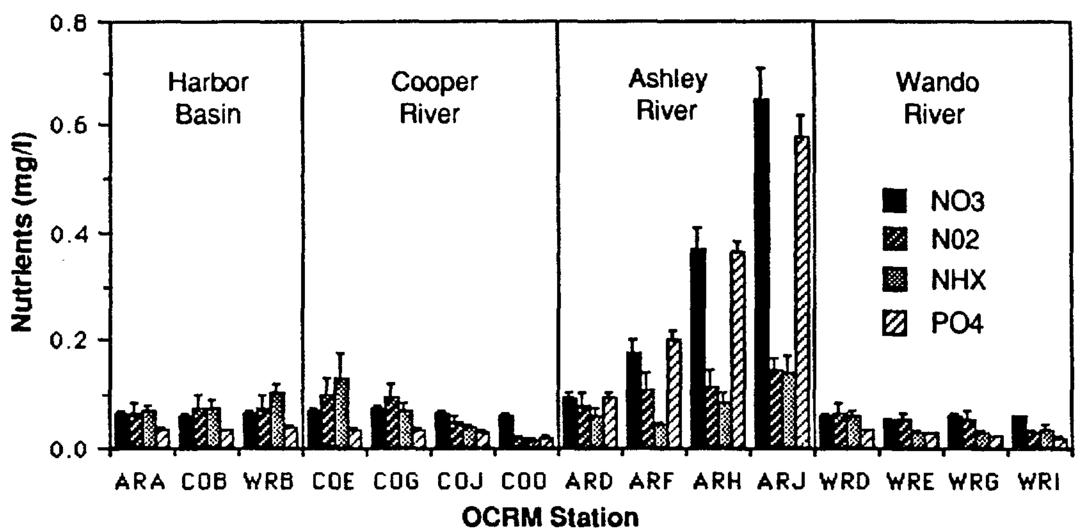

Figure III.23. Annual mean nitrate, nitrite, total ammonia and ortho-phosphate concentrations at hydrographic sampling stations in the Charleston Harbor estuary for 1988. Each value is the mean of 12 surface and bottom values collected between January and December, 1988. Error bars are +/- 1 standard error.

Ranges of values for nitrates, nitrites, ammonia and phosphates observed at each station are presented in Appendix III.2. The 1988 annual mean nitrate value was 0.13 mg/l, while nitrite values averaged 0.07 mg/l, ammonia values averaged 0.02 mg/l, and ortho-phosphate values averaged 0.10 mg/l. Prior to rediversion, the SCDHEC monitoring program recorded ranges of kjeldahl nitrogen between 0.04 and 19.90 mg/l, of nitrate-nitrite between 0.00 and 6.65 mg/l, of ortho-phosphate between 0.00 and 1.56 mg/l, of total phosphate between 0.02 and 4.60 mg/l, and of total ammonia between 0.02 and 13.00 mg/l. The Estuarine Survey Study reported mean concentrations for the entire estuary of 0.053 mg/l nitrate, 0.003 mg/l nitrite, and 0.027 mg/l ortho-phosphate.

Annual mean nitrate and phosphate values at the upstream Ashley River station were five times those found in the other systems, and decreased in a seaward direction (Figure III.23). Statistical analyses (ANOVA) demonstrated that stations ARH and ARJ exhibited significantly higher mean values of nitrate than all other stations in the estuary. In addition, all Ashley River stations demonstrated significantly higher ortho-phosphate values than all other stations in the estuary. In contrast with the Ashley River, the Cooper River appeared to exhibit a slight decrease in mean nitrite and ammonia concentrations in the upstream direction, but the differences were not statistically significant. In addition, the mean ammonia concentrations in the lower Cooper River and upper Ashley River appeared to be slightly higher than in other areas of the estuary, but the differences were not statistically significant.

Comparisons of 1973-1978 nutrient data from the Estuarine Survey Study with 1988 nutrient data indicated similar concentrations (no significant differences) at comparable stations for nitrates and phosphates (Figure III.23). The Estuarine Survey Study data for nitrites, however, were consistently an order of magnitude lower than 1988 values, but this is believed to be an artifact of the method utilized in the Estuarine Survey Study analyses.

Seasonal trends in nutrient concentrations during the 1988 intensive sampling period were not as pronounced as geographic trends, and each basin exhibited unique seasonal fluctuations. The mean, combined surface and bottom nitrate concentrations fluctuated a great deal throughout 1988, and no distinct patterns were discernable. Nitrate concentrations were highest during July in the harbor basin, and Wando and Cooper Rivers, while they were highest during January in the Ashley River (Figure III.24). Ortho-phosphate concentrations exhibited a smoother seasonal pattern with generally higher concentrations being found in the summer months when compared with the rest of the year. Ortho-phosphate concentrations peaked during the period July-September in all systems except the Ashley River which exhibited lower concentrations during September. The Ashley River also demonstrated larger fluctuations in ortho-phosphate concentrations throughout the year than did the other systems. Nitrite concentrations were higher during the summer months in all four areas of the estuary when compared with the rest of the year, and peak values occurred in August or September in the harbor basin and Cooper and

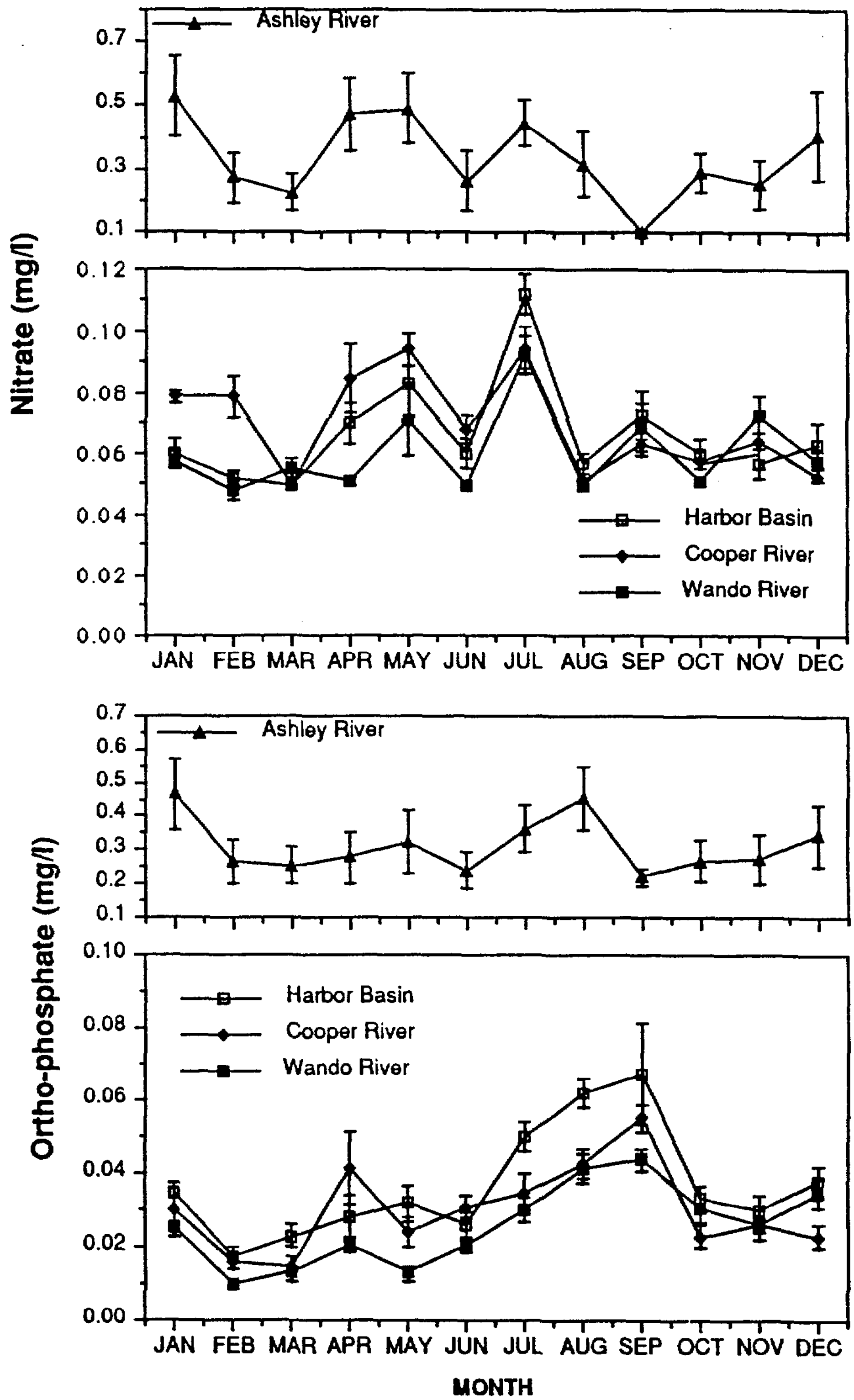

Figure III.24. Seasonal trends in nitrate and ortho-phosphate in the 4 basins of the Charleston Harbor estuary during 1988. Ashley River values were plotted on a different scale to accommodate high values. The error bars are +/- 1 standard error of the mean.

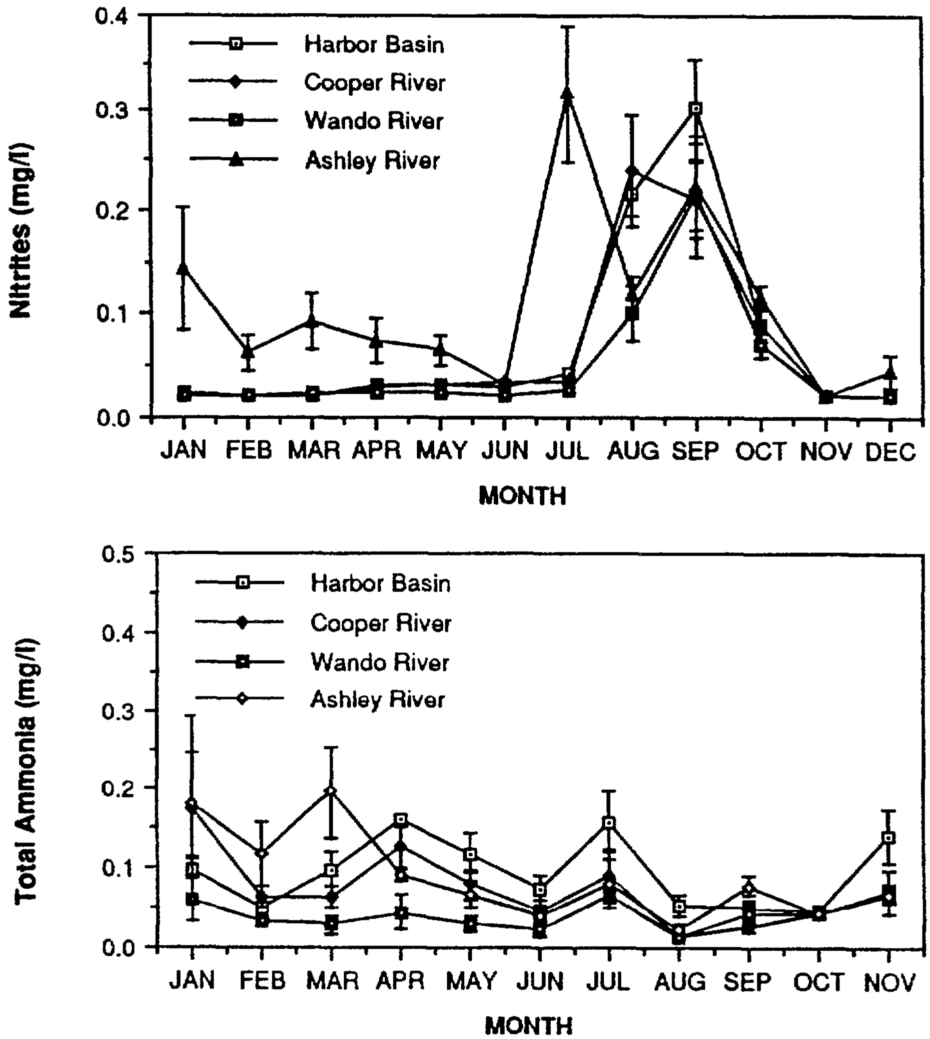

Figure III.25. Seasonal trends in nitrite and total ammonia in the 4 basins of the Charleston Harbor estuary during 1988. Error bars are +/- 1 standard error of the mean.

Wando Rivers (Figure III.25). The Ashley River, on the other hand, exhibited a peak value in July when the other 3 systems exhibited very low nitrite values. As with the orthophosphate values, the Ashley River exhibited much larger fluctuations in nitrite values throughout the year than did the other systems. Values for ammonia concentrations fluctuated throughout the year in all basins, and exhibited no discernable seasonal trends (Figure III.25). This is most likely attributable to the transient nature of ammonia in the estuary as well as large diurnal fluctuations throughout much of the year.

SUMMARY

1. Study objectives of the hydrographic survey conducted by the SCWMRD between November, 1984 and December, 1988 were to: (1) document the seasonal and annual variability in hydrographic conditions in the estuary, (2) compare hydrographic conditions among the three river systems and the harbor basin, and (3) document the changes in hydrographic conditions brought about by rediversion. Hydrographic sampling included the collection of data from surface and bottom during trawl and grab sampling, as well as during high and low tide hydrographic sampling transects. Hydrographic parameters recorded included temperature, specific conductance, salinity and dissolved oxygen at all stations; turbidity, nitrate, nitrite, total ammonia and ortho-phosphate at selected stations.

2. The salinity regimes in the Cooper River and harbor basin were much more saline following rediversion than before. The mean surface freshwater line (<0.5 ppt) was approximately 6 km further upriver in the Cooper River after rediversion, and approximately 2.5 km further upriver on the bottom. It was also apparent that salinities in the harbor basin and Cooper River were primarily controlled by the tidal stage rather than freshwater flow after rediversion. Turbidity levels at the mouth of the harbor were significantly lower during post-rediversion sampling (approximately 3x lower), although no significant differences were observed in the rest of the estuary. Finally, no significant differences were observed for nutrient levels in the estuary between pre- and post-rediversion sampling.

3. Salinity regimes throughout the estuary exhibited no distinct seasonal trends. The salinity regimes in the Ashley and Wando Rivers were less stratified than in the Cooper River and harbor basin. Dissolved oxygen percent saturation exhibited distinct seasonal trends throughout the estuary with highest levels occurring in the winter, lower levels occurred in the spring and fall, and the lowest levels occurred in late summer. Dissolved oxygen levels generally decreased in the upriver direction in the Ashley and Wando Rivers, but in the Cooper River, levels decreased in the upriver direction to the middle stations, and then began to increase in this direction. In addition, the upper Ashley River stations exhibited significantly lower dissolved oxygen percent saturation levels than those found in the harbor basin. Turbidity levels were highest in the upper Ashley River, somewhat lower in the harbor basin and lowest in the Cooper and Wando Rivers. No seasonal trends in turbidity were observed at any stations in the estuary. Nutrient levels were generally higher during summer months than during winter months, although each basin exhibited unique

seasonal changes. Levels of nutrients were similar in the Cooper and Wando Rivers and harbor basin. Nutrient levels in the Ashley River, however, were significantly higher than nutrient levels recorded from the rest of the estuary. The upper Ashley River stations exhibited extremely high concentrations of nitrate and orthophosphate (often 5-10x higher than rest of estuary) which decreased in the seaward direction.

CHAPTER IV

ORGANIC CARBON AND NUTRIENT DYNAMICS

by

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INTRODUCTION

The distribution and movements of organic matter and nutrient elements along an estuarine gradient reflect both natural biogeochemical processes as well as human influence. Changing ratios of carbon, nitrogen and phosphorus can indicate degrees of nutrient limitation and excess (Welch, 1980) and can also be used to deduce dominant properties of estuarine metabolism and net organic production (Nixon and Pilson, 1984; Smith *et al.*, 1987). Many symptoms of declining water quality in estuarine systems (such as noxious algal blooms, elimination of desirable species, and oxygen depletion) are related to changes in the loading and distribution of critical nutrient fractions (Ketchum, 1969; Ryther and Dunstan, 1971) in interaction with patterns of river flow and estuarine hydrodynamics (Stanley, 1987).

Concurrent with human development are changes in land use with increases in point source and non-point source nutrient loading. In addition to increased loading, coastal development often involves changes in freshwater discharge, estuarine hydrology, and wetland coverage. All of these factors modify basic ecological processes which maintain nutrient balances and water quality, as summarized below.

Natural Patterns of Estuarine Nutrient Distribution:

Nutrient dynamics in estuarine systems reflect interactions between riverine and oceanic driving forces superimposed on effects of internal processes and wetland exchanges. Estuarine gradients between riverine and ocean influences interact with ecologic factors related to primary production, respiration, and decomposition in diverse open water and wetland habitats to produce complex patterns in nutrient distribution and flux. Natural sources of nutrient fractions along estuarine reaches result from biogenic remineralization, gaseous fixation (nitrogen fixation and photosynthetic C fixation), and resuspension of benthic sediments. Nutrients are removed from estuarine waters by biogenic uptake, sedimentation, flocculation, and gaseous export (denitrification and respiratory CO_2 release). The relative roles of these processes change temporally and spatially within an estuarine system and are modified by terrestrial runoff and by tidal exchanges with bordering wetlands.

Wetland Exchanges - Nutrient exchange with tidal wetlands may represent an important factor in estuarine nutrient distributions and fluxes. Background information on nutrient flux through coastal wetlands has increased considerably in the last 10 years (see review by Nixon, 1980). Nutrient cycling and productivity in wetland ecosystems are controlled by complex interactions of biology, geochemistry and hydrology (Gosselink and Turner, 1978). Many intertidal marshes tend to import particulate matter and export dissolved fractions of nitrogen, phosphorus, and organic carbon to estuarine and coastal water. (Valiela *et al.*, 1978; Woodwell and Whitney, 1977; Woodwell *et al.*, 1979; Jordan *et. al.*, 1983).

Recent studies of the North Inlet salt marsh in South Carolina documented considerable nutrient export from a complex 34 km^2 marsh-estuarine ecosystem (Kjerfve and McKellar, 1980; Whiting *et al.*, 1987). Mechanisms controlling this export are related to the nutrient dynamics of the dominant vegetation (Hopkinson and Schubauer, 1984; Whiting *et al.*, 1989) and high rates of decomposition and remineralization in marsh substrates (Pomeroy and Wiegert, 1981) coupled with the diffusion and drainage of substrate pore waters during low tide exposure (Gardner, 1975; Wolaver *et al.*, 1980, 1983; Whiting *et al.*, 1989). Impounded wetlands, which occupy considerable portions of many estuarine areas, tend to retain more nutrients and organic matter that open tidal marshes largely because of altered hydrology and vegetative growth patterns (McKellar and Marshall, 1984; McKellar and Kelley, 1987).

Our knowledge of seasonal dynamics of southeastern tidal freshwater wetlands is less defined than that for southeastern salt marshes. Water velocity reduction by freshwater wetland vegetation results in sediment and particulate nutrient deposition. Wetland community composition and annual structural integrity greatly influence the filtering capacity (Allen, 1978). Boto and Patrick (1979) found certain coastal wetlands import solid material during some seasons and export the same material during other seasons. Freshwater wetlands often remove dissolved nutrients from the water during the growing

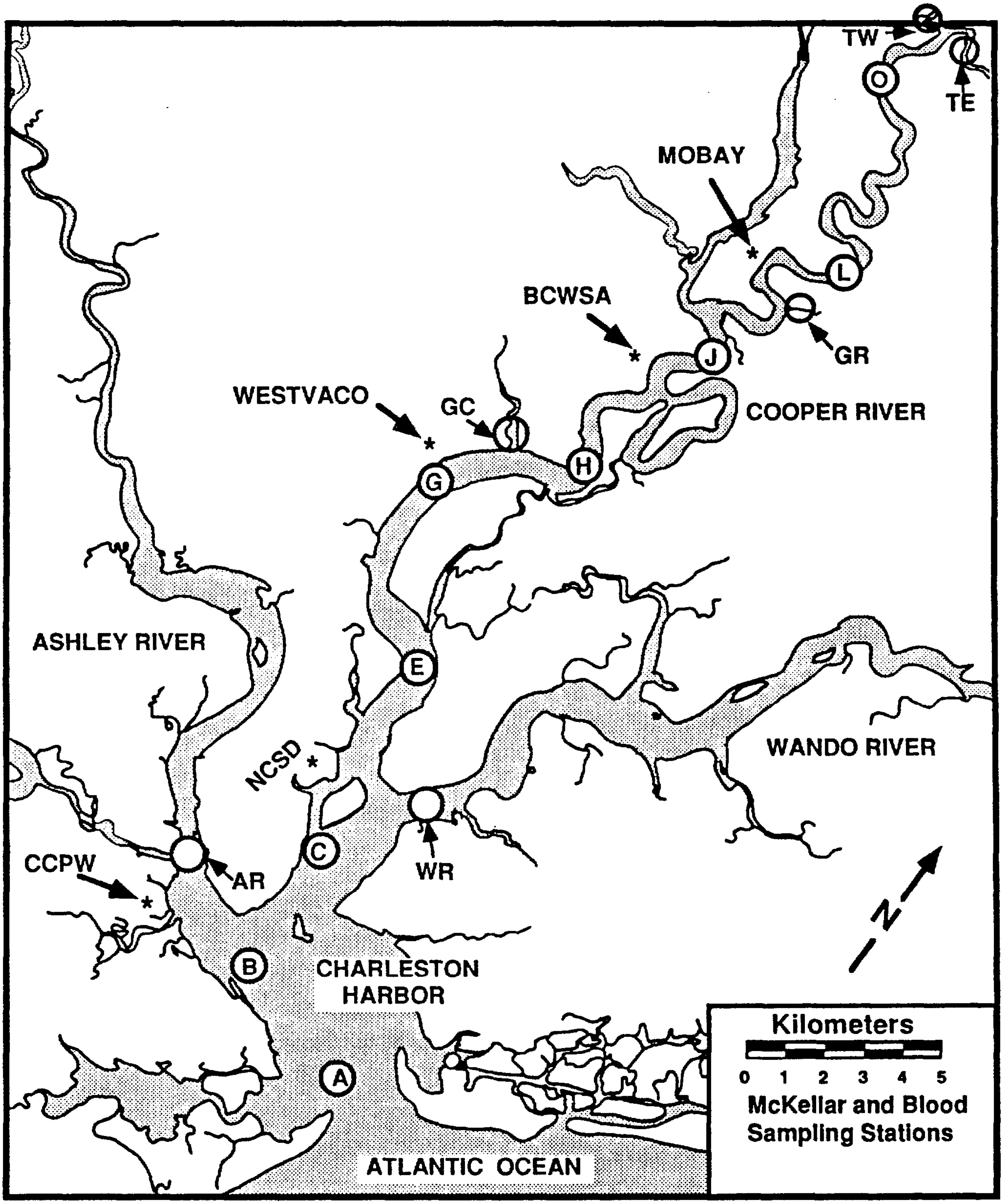


Figure IV.1. Sampling station locations (indicated by letters). Tributary stations are Ashley River (AR), Wando River (WR), Goose Creek (GC), Grove Creek (GR), and East Branch of the Cooper River (TE). Major point source locations are indicated by * and include municipal: Central Charleston Public Works (CCPW), North Charleston Sewer District (NCSD), and Berkeley County Water and Sewer Authority (BCWSA), and industrial: Westvaco Paper Mill (WESTVACO) and Mobay Chemical (MOBAY). Permitted discharges for BOD and ammonia are listed in Table IV.1.

Table IV.1. Point Source Permits for BOD and Ammonia Discharge to the Cooper River and Charleston Harbor.

SOURCE	River Km	Discharge (MGD)	Load BOD	(lbs/day) Ammonia
Charleston Comm. Public Works	7	18	4,504	3,002
North Charleston Sewer Distr.	11	18	4,504	31,825
Westvaco Paper Mill	19	20	13,014	-----
Berkley Co. Water and Sew. Author.	25	10	2,502	1,668
MOBAY Chemical	32	6.5	1,985	545
DuPont	47	1.2	420	-----

Chemical Analyses:

Particulate matter collected on the filters was analyzed for total particulate organic carbon (POC), representing an aggregated sum of fine suspended detritus and plankton biomass (phytoplankton, zooplankton and bacteria). Filters for POC analysis were folded and frozen in foil envelopes for storage until analysis. POC was determined by dry combustion of material on the filters followed by CO_2 analysis by infrared absorption using an Oceanography International (OI) carbon analyzer. Additional filters were analyzed for chlorophyll-a (corrected for phaeopigments) as an indicator of viable phytoplankton biomass. Filters for chlorophyll analysis were frozen under 1 ml saturated $MgCO_3$ solution until they were analyzed by standard fluorometric analysis before and after acidification, using a freeze-thaw acetone extraction procedure (Glover and Morris, 1979).

Filtrate was analyzed for dissolved organic carbon (DOC) and inorganic fractions of nitrogen and phosphorus. Samples for DOC analysis (1 ml aliquots) were frozen in pre-combusted glass ampules for storage. DOC was determined by the persulfate oxidation method (Menzel and Vacarro, 1964) followed by infrared CO_2 analysis on the OI carbon analyzer. Dissolved ortho-phosphate (PO_4) was determined by the acid-molybdate method of Murphy and Riley (1962). Ammonium was determined by the hypochlorite method of Solarzano (1969) after preservation with phenol (Degobbis, 1973). Nitrate-nitrite was analyzed by the cadmium reduction technique (APHA, 1976). All nitrogen and phosphorus fractions were analyzed on an Orion Ionanalyzer.

Statistical Analyses:

To facilitate statistical analyses, distributions of carbon, nitrogen, phosphorus, and chlorophyll values were normalized by log transformation. A multi-variate analysis of co-variance was used to determine the significance of the relationships between these parameters and other independent variables including time of year, river flow, tidal stage, salinity, depth, and position within the estuary. Tukey and Bonferroni multiple comparison procedures were used to locate specific differences in the factors (SAS User's Guide 1985).

Specific temporal factors included tidal stage (high or low), month, and season where seasons were defined as Winter (Jan.-March), Spring (Apr.-June), Summer (July-Sept.), and Fall (Oct.-Dec.). Spatial factors included depth (surface or bottom), station, and region where regions were defined as "harbor" (RK 0-10, stations A and C,), "mid-estuarine" (RK 10-25, stations E and G), "oligohaline" (RK 25-35, stations J and L), and "fresh water" (RK >35, station L).

Additionally, the significance of tributary inputs to the main channel water quality was determined by similar statistical comparisons of low tide concentrations at the tributary inlets with concentrations at adjacent stations in the main channel.

RESULTS AND DISCUSSION

Distributions of water quality, organic matter, and nutrients in the Cooper River and Charleston Harbor varied significantly (a ≤ 0.05) with time and spatial position in the estuary. To analyze these patterns of variability, we first present trends in river flow and salinity distributions as a primary determinant of temporal and spatial variability throughout the estuary. Then, the dominant patterns of distribution of each major constituent of water quality (turbidity and dissolved oxygen), organic carbon, and nutrients are examined in detail. Statistical results for evaluating the significance of observed differences and correlations are provided in Appendix IV.A and IV.B.

River Flow and Salinity Distributions:

Daily discharge in the Cooper River was extremely variable during the study period ranging from 0 m^3/s to 334 m^3/s (Figure IV.2). Although daily fluctuations due to hydroelectric power generation at Pinopolis Dam were large, seasonal variability was more

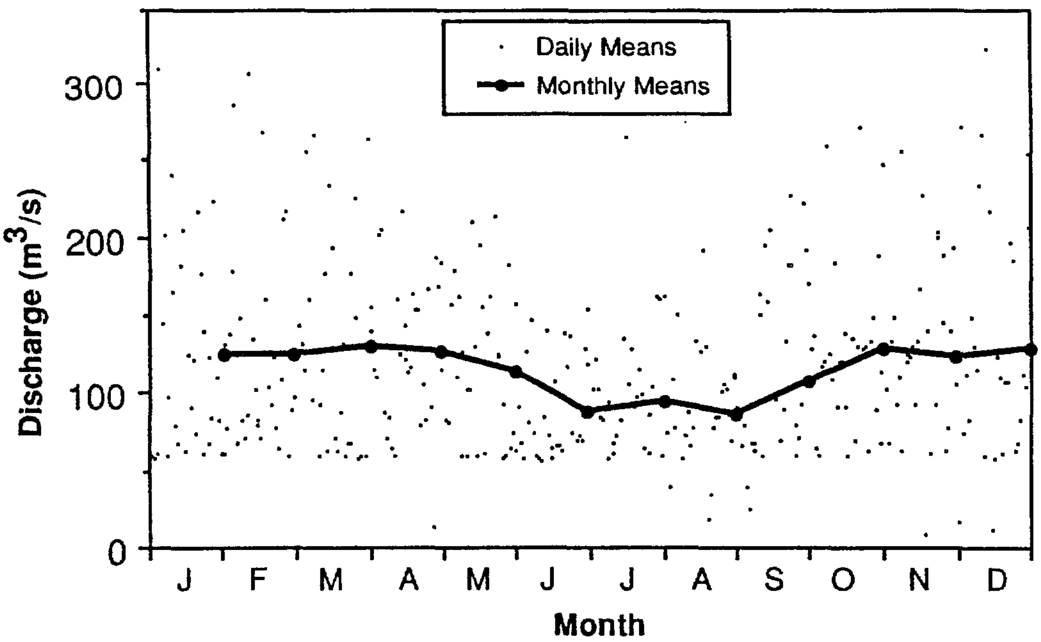


Figure IV.2. Discharge (cubic meters per second) in the Cooper River.

moderate. Monthly mean flows remained relatively constant throughout most of the year (120-140 m^3 /s), with a 30-35% decrease during mid to late summer when flows were 80-90 m^3 /s. The mean discharge during the entire study period was 117 m^3 /s (4130 cfs). This represents a 71% decrease in mean flows in the Cooper River from conditions prior to the 1985 rediversion.

Salinity ranged from fresh water [0-0.5 ppt above river kilometer (RK) 45] to >30 ppt at the mouth of Charleston Harbor (Figure IV.3). There was an approximate linear decline in salinity with distance between the harbor and RK 35 (mean slope of about 0.8 ppt/RK) suggesting no major discontinuities in salt distributions through the estuary. An approximate transition zone between fresh and brackish water occurred around RK 35 (Sta.L, between Mobay Chemical and General Dynamics) where low tide salinities were typically fresh (<0.5 ppt) and high tide salinities sometimes reached 6 ppt (mean = 3.4 $\pm$ 0.7 ppt in bottom waters). Salinities >0.5 ppt were rarely observed beyond RK 39 (near the SCEG power plant), indicating the landward extent of salt water intrusion during high tide.

The maximum vertical gradient produced by the salt wedge occurred through the mid-estuarine stations (E and G, RK 12-20) where salinities in the bottom waters were

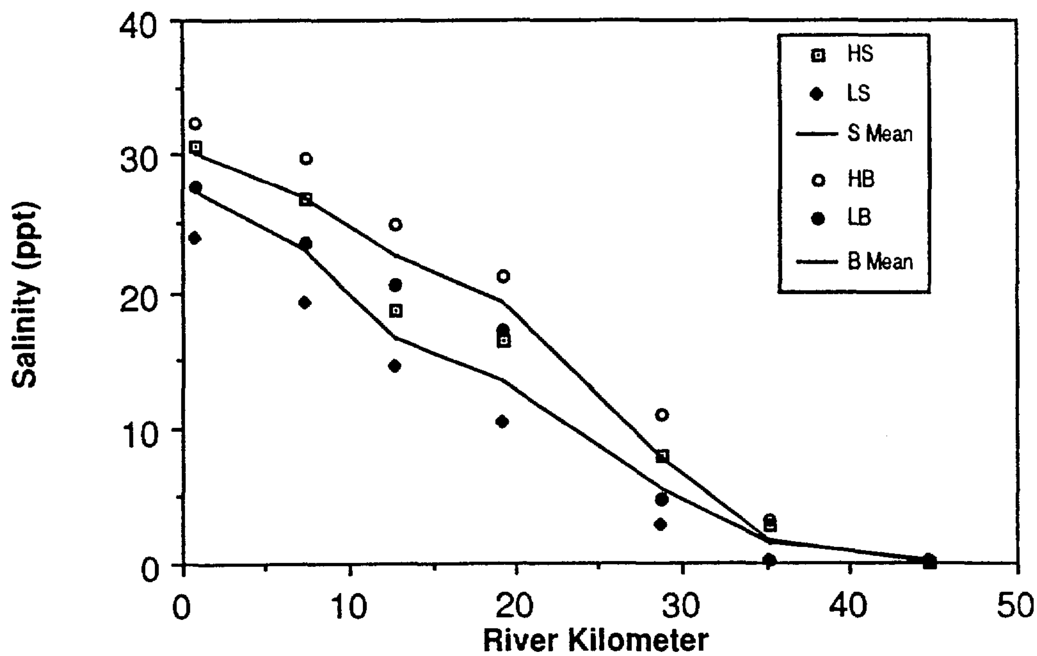


Figure IV.3. Salinity at stations along the estuarine gradient. Data average all months for high tide at the surface (HS), low tide at the surface (LS), mean surface (SMean), high tide bottom (HB), low tide bottom (LB), and mean bottom (BMean).

usually 5 to 7 ppt greater than in the surface water (Figure IV.3). This area of maximum vertical stratification corresponded to the region from the Navy Ship Yard to Goose Creek.

Temporal correlations between river discharge and salinity distributions were weak, although significant for some stations and river flow conditions (Table IV.2). The strongest and most significant correlations were found for weekly mean flows prior to sampling. Table IV.2 indicates that 37-56% of the salinity variations in the bottom waters throughout the estuary could be explained by the weekly mean river flow prior to sampling. This analysis suggested that salt distributions (as well as other aspects of water quality) reflected the integrated influence of river flow over the previous week. Maximum discharges during the week prior to sampling also explained some of the variations in salt distribution, although these correlations were not as consistent for all of the stations (Table IV.2). Clearly, much of the variation in salt distribution was controlled by factors other than river discharge (such as wind speed and direction and the magnitude of ocean tides).

Table IV.2. Pearson Correlation Coefficients for river discharge and low-tide salinities.

Station	Sampling Day Flow	2-Day Mean Flow	3-Day Mean Flow	7-Day Mean Flow	7-Day Max Flow
A	-.14[a]/.18[b]	-.40/-.39	-.25/-.32	-.50/-.39	-.39/-.32
C	.28/-.05	-.06/-.49	-.45/-.46	-.30/-.75	-.32/-.71
E	.29/-.07	.03/-.41	-.36/-.29	-.30/-.61	-.37/-.33
G	.14/-.17	-.21/-.54	-.57/-.36	-.59*/-.65*	-.68*/-.56
J	.29/-.07	-.27/-.59	-.52/-.63*	-.40/-.74*	-.46/-.63*
L	.41/-.41	-.46/-.46	-.49/-.49	-.64*/-.64*	-.51/-.51

[a] surface
[b] bottom
* statistically significant at alpha = .05

Turbidity and Dissolved Oxygen:

Secchi disk observations displayed significant seasonal and spatial trends in the estuary (Appendix IV.A1-IV.A2). Seasonal trends suggested significant turbidity peaks (low secchi disk observations, 0.5-0.9 m) in March and September (Table IV.3, Figure IV.4). corresponding to winter and summer peaks in particulate organic carbon and phytoplankton biomass (see following sections, Figures IV.8, IV.11). The maximum water clarity (secchi disk values 1.1-1.2m) occurred in the late spring (June) corresponding to lower phytoplankton concentrations in the mid-estuarine and oligohaline reaches of the estuary. The general correlations among secchi disk observations, POC, and chlorophyll were significant but explained only 5%-10% of the total variability in these parameters (Appendix IV.B1).

Spatial trends in secchi disk observations indicated turbidity peaks (low secchi disk values) in the oligohaline reaches (RK 30-35) and in the harbor region, RK 0-8 (Figure IV.5, Appendix IV.A2). The turbidity maximum at the oligohaline region is typical in other estuaries (Fisher *et al.*, 1988) where flocculation of particulate matter typically occurs at the interface between fresh and brackish waters. The high turbidity in the harbor region was perhaps related to the influx of turbid water from the Ashley River which exhibited significantly lower secchi disk depths (by 20-36 cm) than the adjacent harbor stations (Table IV.4). In general, drainage from the more developed tributaries (Ashley River and Goose Creek) showed significantly lower secchi disk depths (higher turbidities) than adjacent estuarine waters (Table IV.4), correlating with higher concentrations of particulate organic

Table IV.3. Secchi Disk depth means by season, tide, and region.

		Mean Depth (cm)	Std. Error
Season:	Winter	96.3	7.0
	Spring	108.0	5.7
	Summer	85.0	5.7
	Fall	101.0	6.1
Tide:	High	99.1	2.7
	Low	92.7	2.3
Region:	Harbor	85.0	4.7
	Mid-estuary	110.0	4.6
	Oligohaline	87.4	4.5
	Fresh Water	104.7	4.5

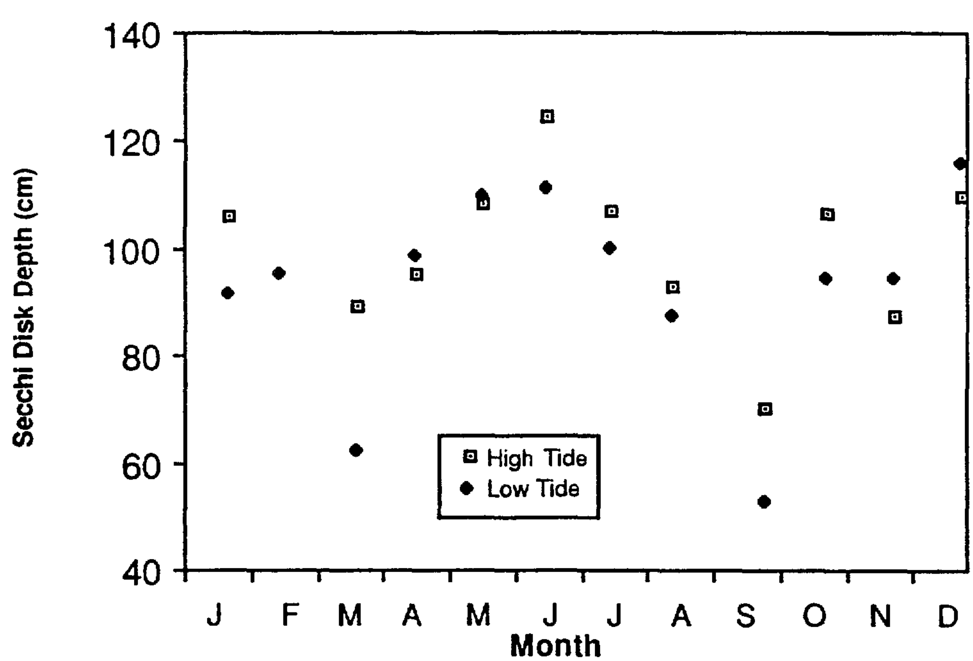


Figure IV.4. Mean monthly secchi disk depth from the surface averaging all stations along the estuarine gradient for high and low tide.

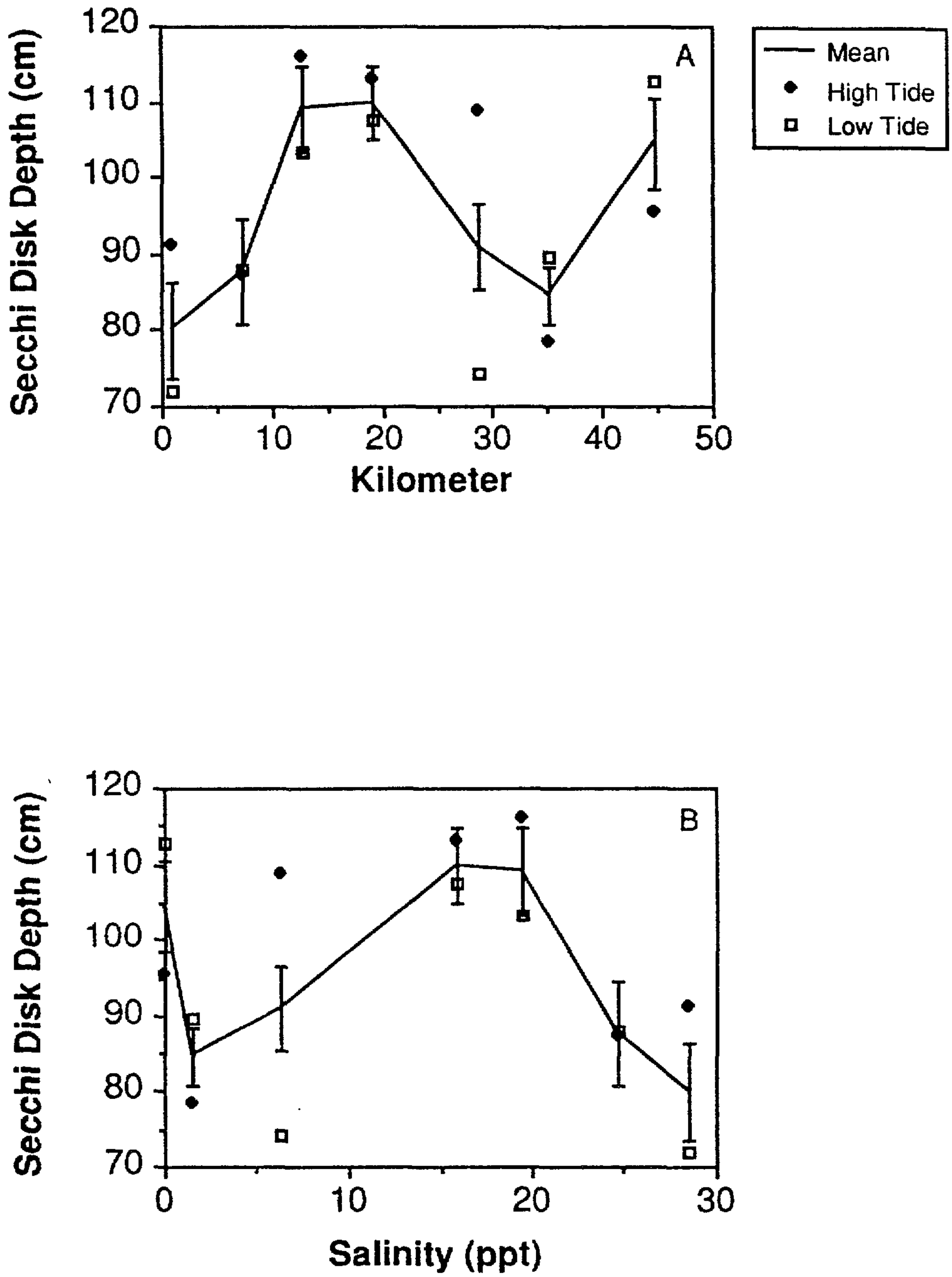


Figure IV.5. Secchi disk depth (cm) from the surface along the estuarine gradient (A) and versus mean station salinity (B). Means average high tide, low tide and all data at each location.

Table IV.4. Secchi Disk Differences (cm) between tributary inputs and main channel. (Low tide means averaged through the water column over the entire study period, n = 21-22, $\bar{X}t$ = tributary mean, $\bar{X}c$ = main channel mean, * indicates differences, p≤.05.)

TRIBUTARY	$\bar{X}t$	±SE	MAIN CHANNEL STA	$\bar{X}c$	±SE	Difference ($\bar{X}t$-$\bar{X}c$)
East Branch	122	4	TW	123	4	-1
Grove Creek	77	4	J	74	4	3
			L	90	4	-13*
Goose Creek	78	3	G	108	3	-30*
			H	86	3	-8
Wando River	98	5	C	88	5	10
			E	103	5	-5
Ashley River	52	6	A	72	6	-20*
			C	88	5	-36*

matter (POC) and chlorophyll in these inputs to the estuary (Table IV.7 and IV.8). Other factors affecting harbor turbidity could have been related to harbor dredging, shipping activities and wind-driven disturbance.

Dissolved oxygen in the estuary varied significantly with season, position within the estuary, and depth (Appendix IV.A3-IV.A4). Seasonal variability through the water column (Table IV.5, Figure IV.6), reflected a strong negative correlation with temperature (Appendix IV.B1), indicating temperature effects on oxygen saturation as well as community respiration. The lowest oxygen concentrations (4-5 mg/l) occurred during the warmest months (Figure IV.6) in the bottom waters of the mid-estuarine region (Figure IV.7, Appendix IV.A4). In August, oxygen concentrations remained below 5 mg/l for both low tide and high tide samplings. Although no violations of Class SC standards (4 mg/l) were observed, our readings were probably higher than minimum values since all samplings were during daytime periods. Nighttime community respiration could yield lower concentrations than observed, causing frequent water quality violations during the warmest months.

Spatial variability in dissolved oxygen concentrations was dominated by a steady decline from freshwater reaches through the oligohaline zone reaching minimum values in the mid-estuarine area and upper harbor (RK 5-20; Figure IV.7, Appendix IV.A4). This

Table IV.5. Dissolved oxygen means by season, tide, region, and depth.

		Mean DO (mg/l)	Std. Error
Season:	Winter	9.7	0.2
	Spring	6.8	0.2
	Summer	5.5	0.2
	Fall	7.0	0.2
Tide:	High	6.9	0.1
	Low	7.1	0.1
Region:	Harbor	7.1	0.1
	Mid-estuary	7.0	0.1
	Oligohaline	7.4	0.1
	Fresh Water	7.8	0.1
Depth:	Surface	7.1	0.1
	Bottom	6.9	0.1

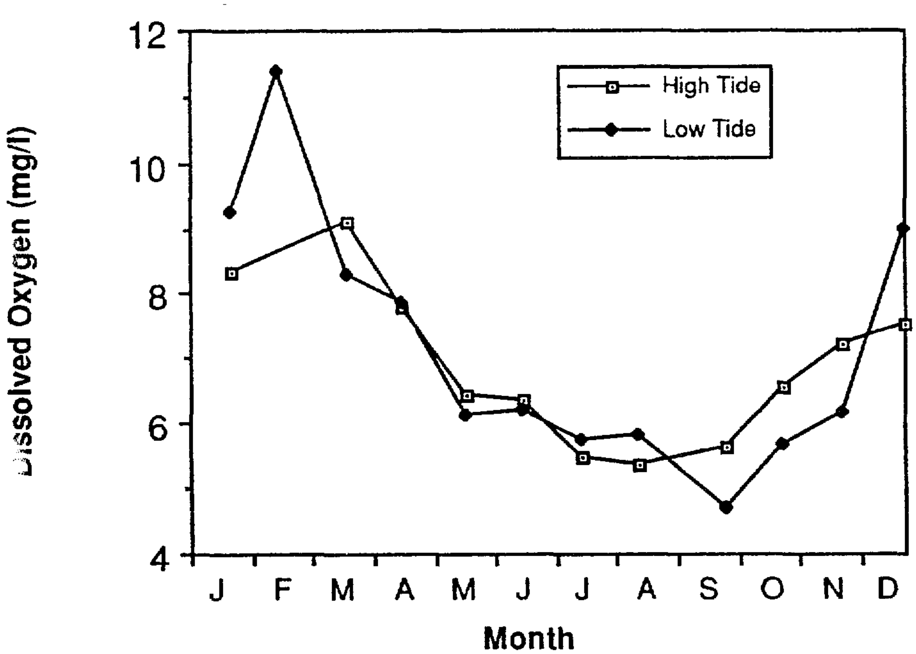
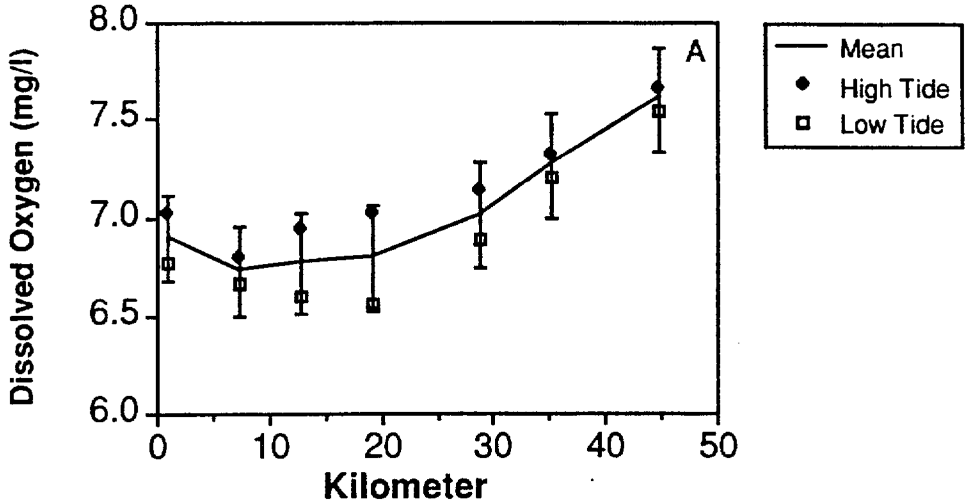


Figure IV.6. Mean monthly dissolved oxygen (mg/l) for high and low tide averaging all stations along the estuarine gradient of the Cooper River.



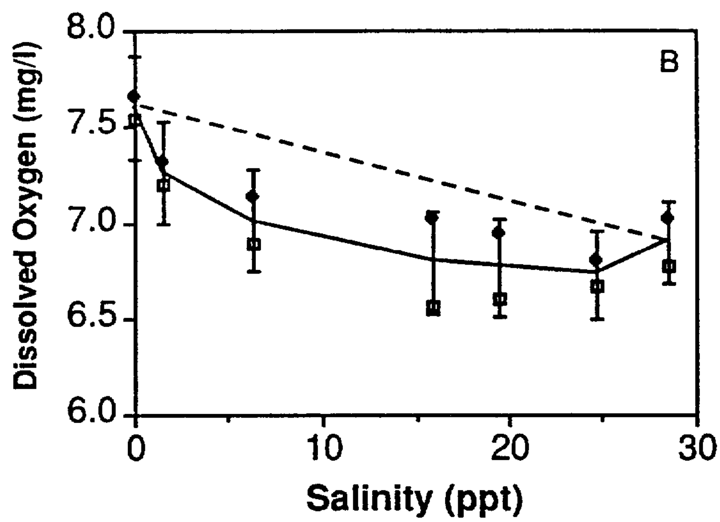


Figure IV.7. Dissolved oxygen (mg/l) from the surface along the estuarine gradient (A) and versus mean station salinity (B). Means are averages of high and low tide for all data at each location.

apparent oxygen sag yielded maximum oxygen deficits of 18-25% suggesting higher rates of oxygen demand in this region of the estuary. This pattern was probably related to the large point sources of BOD discharge to this region of the estuary (see Table IV.1). The Westvaco unbleached paper mill was permitted to discharge >13,000 lb/day carbonaceous BOD at RK 20, and the North Charleston Sewer District (NCSD) was permitted to discharge treated municipal and industrial process wastes amounting to >30,000 lb/day of ammonia and its related nitrogenous oxygen demand (RK 11).

The spatial variability of oxygen across the estuarine salinity gradient was not significantly affected by tributary inputs (Table IV.6). Tributary inputs of dissolved oxygen were usually similar to adjacent estuarine water (mean differences <0.5 mg/l).

Table IV.6. Dissolved oxygen differences (mg/l) between tributary inputs and main channel. (Low tide means averaged through the water column over the entire study period, N = 21-22, $\bar{X}t$ = tributary mean, $\bar{X}c$ = main channel mean, * indicates significant differences, $p \leq .05$.)

			MAIN CHANNEL			
TRIBUTARY	$\bar{X}t$	±SE	STA	$\bar{X}c$	±SE	Difference ($\bar{X}t$-$\bar{X}c$)
East Branch	7.9	0.1	TW	8.2	0.1	-0.3*
Grove Creek	7.6	0.1	J	7.4	0.1	0.2*
			L	7.8	0.1	-0.2
Goose Creek	7.1	0.1	G	7.0	0.1	0.1
			H	7.1	0.1	0.0
Wando River	6.9	0.2	C	6.5	0.2	0.4*
			E	6.9	0.2	0.0
Ashley River	6.5	0.1	A	6.7	0.1	0.2
			C	6.5	0.1	0.0

The most pronounced vertical gradient in oxygen concentrations occurred in the mid-estuarine area (RK 10-20, Figure IV.7) coincident with the maximum vertical salt gradient (Figure IV.3). Bottom waters in this area were generally 0.4-0.5 mg/l lower in oxygen concentration than the surface waters indicating decreased aeration from the atmosphere and suggesting considerable rates of oxygen demand in the bottom waters or sediments. Community respiration in the bottom waters could be enhanced by subsurface discharges of BOD (Westvaco, Mobay, and NCSD, Figure IV.1, Table IV.2). Furthermore, decaying organic matter sinking from the surface waters could also add to the oxygen demand of the bottom waters. This mechanism was clearly suggested by significantly higher concentrations of particulate organic carbon in the bottom waters during all seasons and over the entire study area (see next section, Figures IV.9, IV.10, and Appendix IV.A5).

Organic Carbon:

Organic carbon in estuarine water is composed of both particulate and dissolved material. Particulate organic carbon (POC) represents biomass of planktonic organisms (phytoplankton, zooplankton, bacteria) as well as detrital particles. POC is derived from primary production and senescence of planktonic biomass as well as from detrital export from contributing watersheds and wetlands. Dissolved organic carbon (DOC) represents a wide variety of excretion products and leachates from decaying detritus and is also derived from internal water column processes (excretion and decay) plus watershed export and wetland exchange.

Particulate Organic Carbon - POC ranged from 0.1 to 5 mg/l with mean values varying largely between 1 and 2 mg/l. Temporal variability was dominated by a bimodal seasonal pattern with significant peaks in both summer and winter (Figures. IV.8 and IV.9, Appendix IV.A5-IV.A6). These peaks contributed to observed peaks in turbidity (Figure IV.4) and were significantly correlated with similar peaks in phytoplankton biomass (Appendix IV.B1, Figures IV.11, IV.12). Using a standard 35:1 ratio of phytoplankton carbon to chlorophyll-a (APHA, 1989), we estimated that POC was generally dominated by detrital material (75-80%) with phytoplankton biomass typically composing 20%-25% of the total. However, the phytoplankton dominated total POC (54%-75%) during the summer in the surface water of mid-estuarine and harbor regions.

Spatial patterns for POC distribution were dominated by distinct vertical gradients in the water column and by longitudinal differences from fresh water to the harbor (Figure IV.9 and IV.10, Appendix IV.A5-IV.A6). Consistently higher concentrations in the bottom waters (0.3-0.5 mg/l higher than surface water) indicated a net accumulation of POC in the lower levels of the estuarine water column. This pattern was observed throughout the year

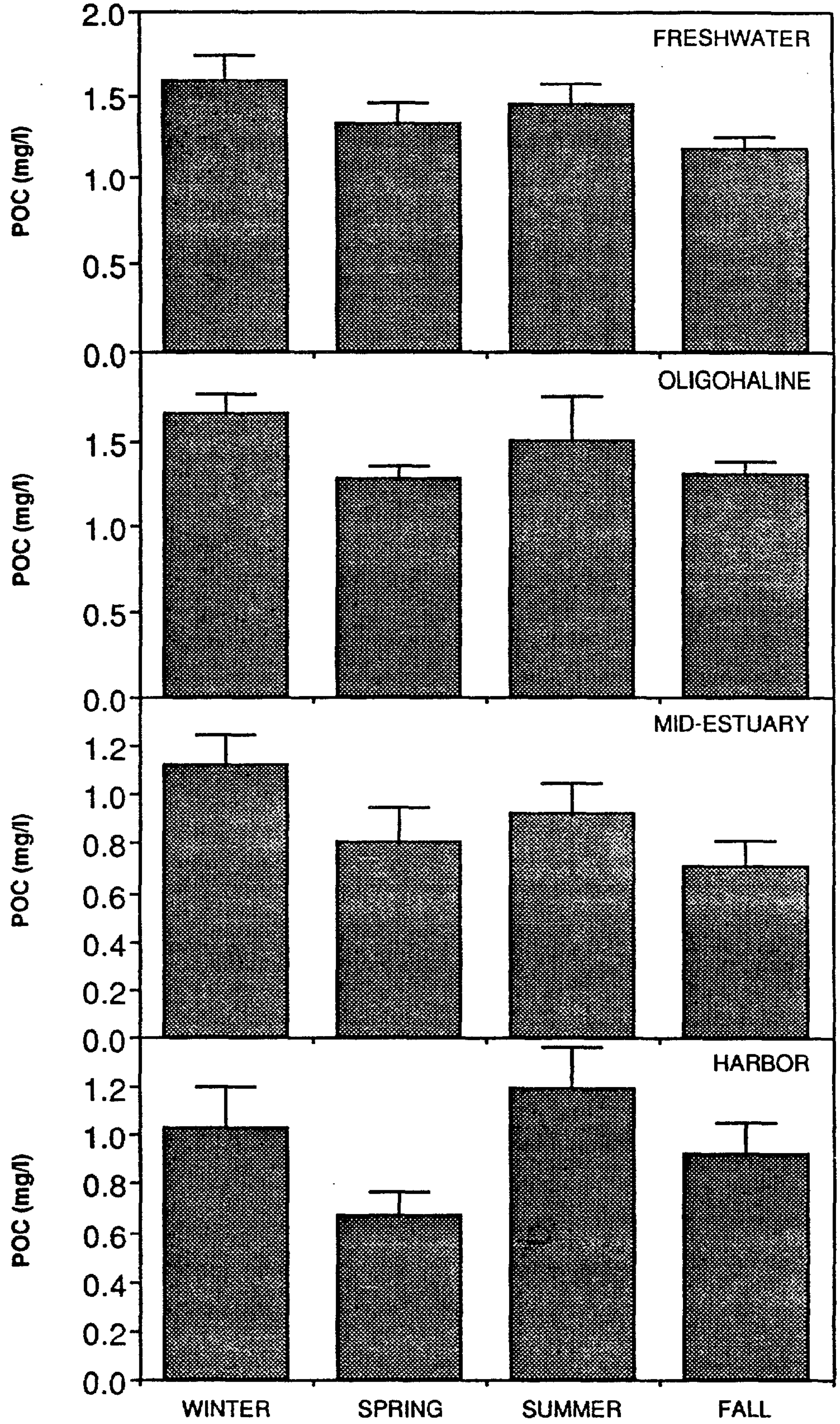


Figure IV.8. Mean seasonal particulate organic carbon (POC) concentration (mg/l) and standard error by estuarine region. Concentrations are averaged overall depths, tides and stations within a given estuarine region.

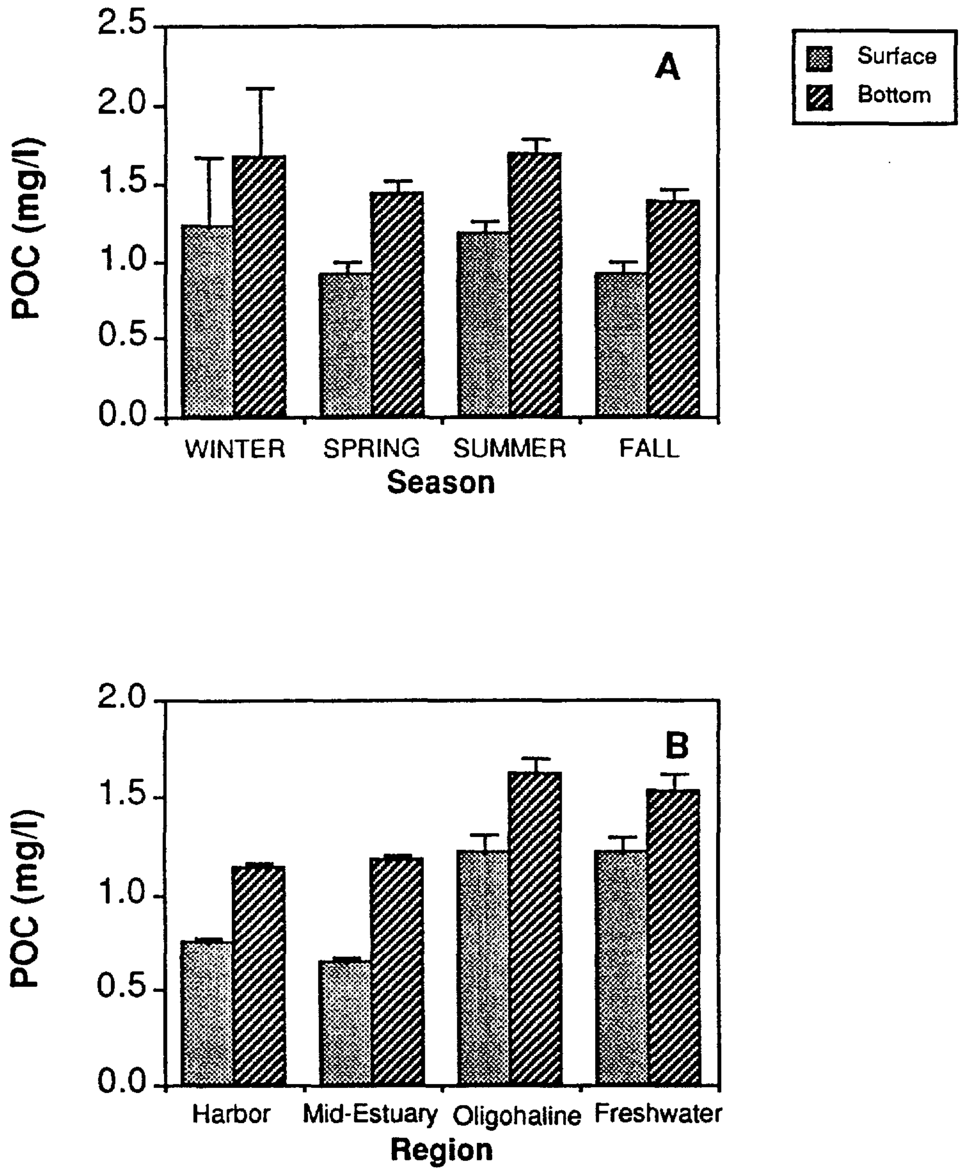


Figure IV.9. Mean particulate organic carbon (POC-mg/l) for surface and bottom waters averaging all months (A) within a season and (B) all stations within a given region. Error bars are standard errors.

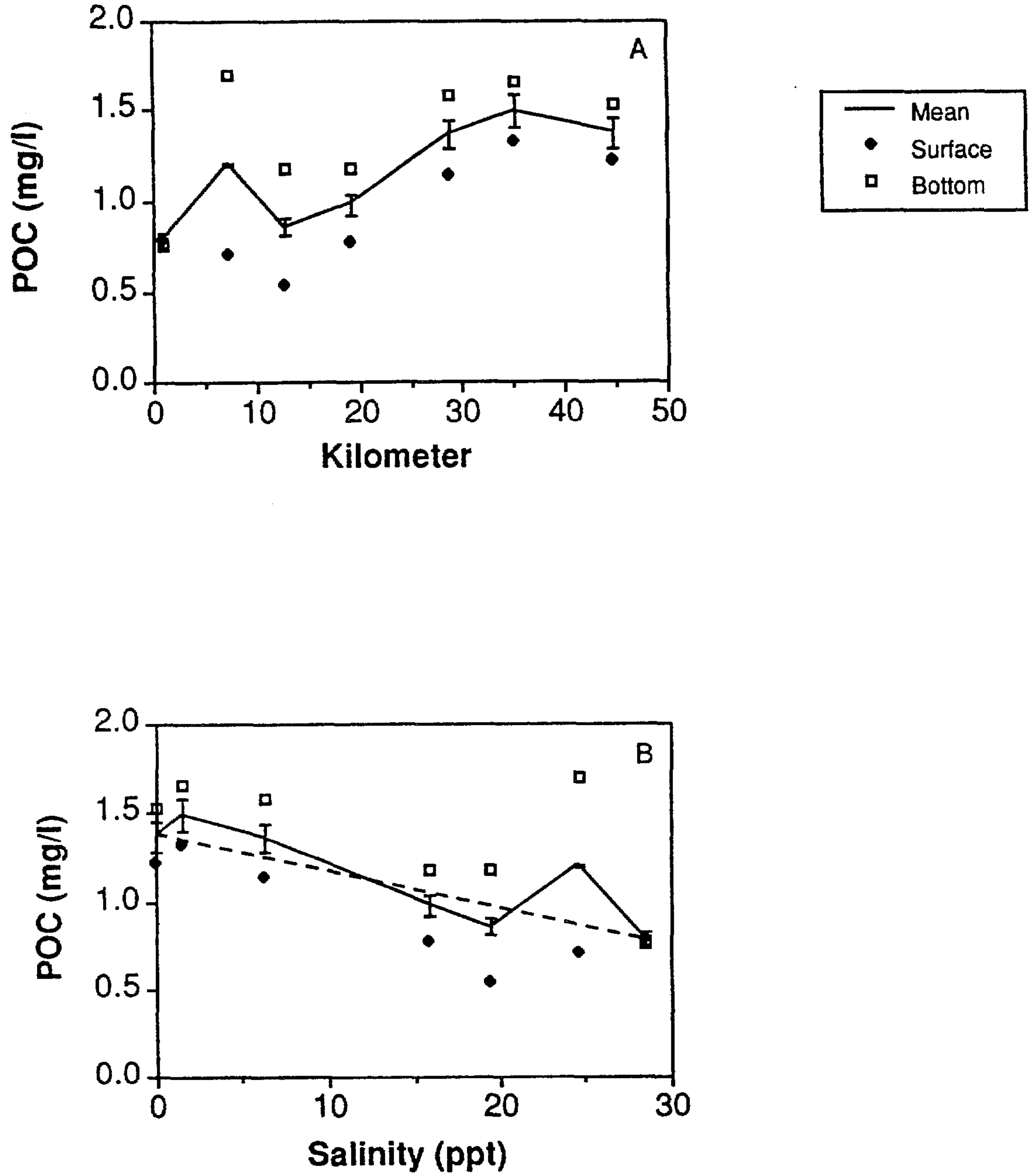


Figure IV.10. Mean particulate organic carbon (POC-mg/l) (A) in surface and bottom waters and overall for each station along the estuarine gradient and (B) versus mean salinity for each station. Error bars are standard errors.

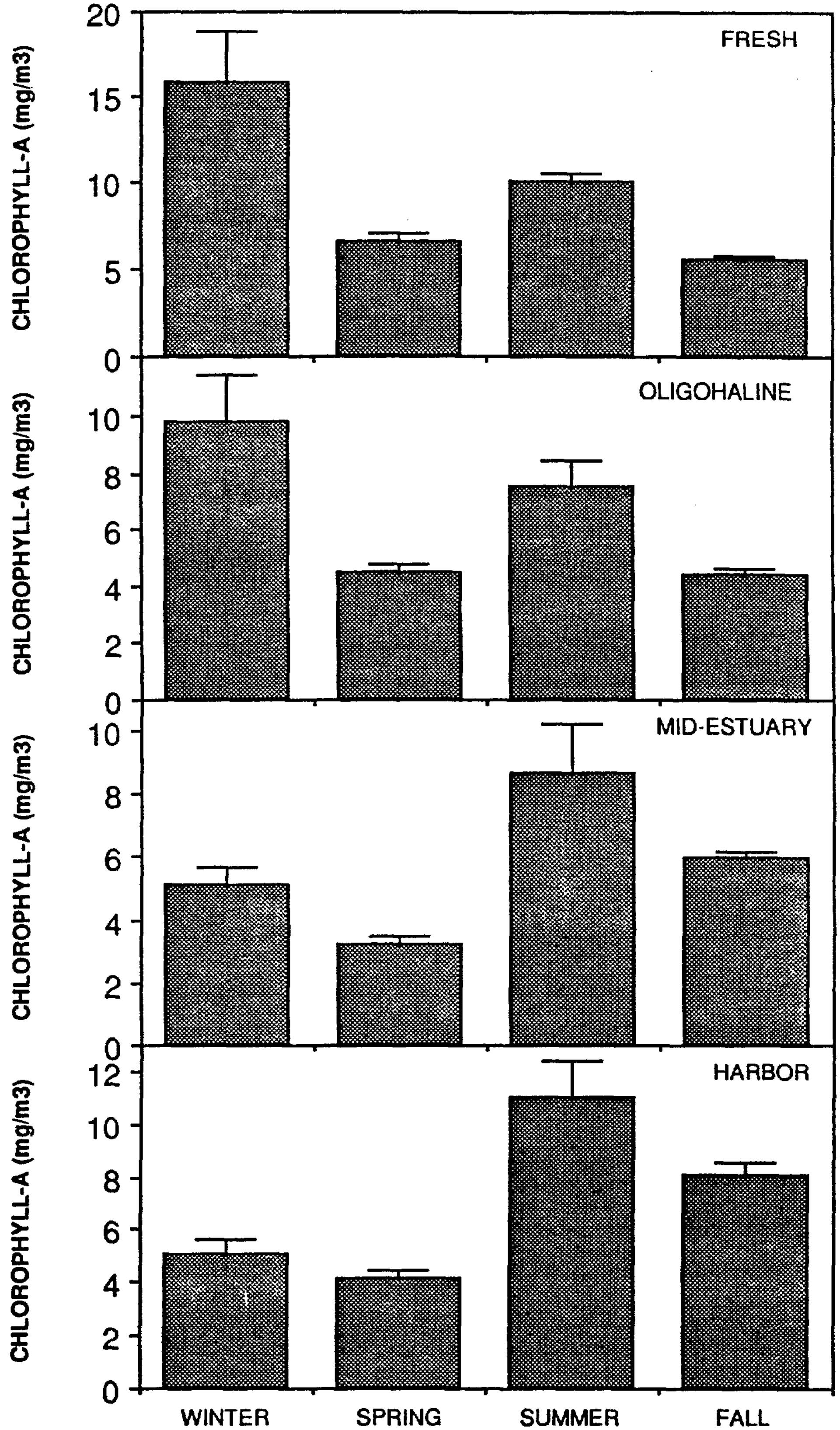
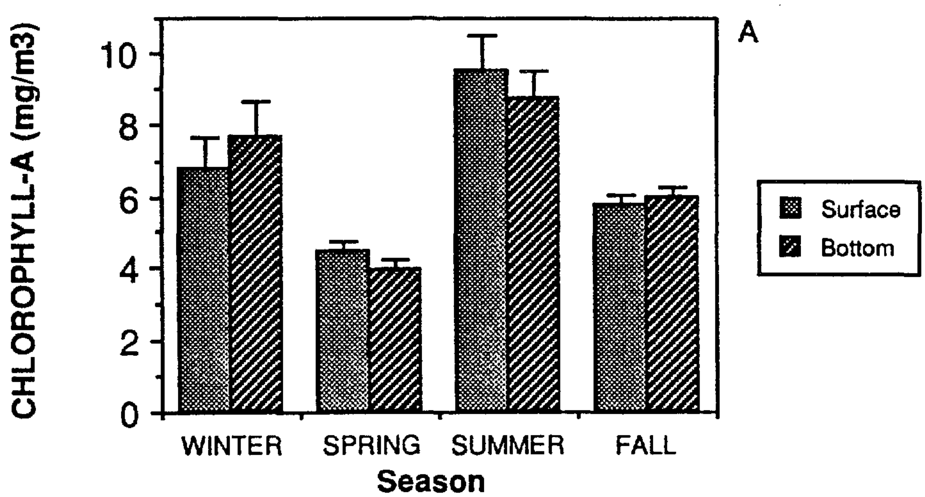


Figure IV.11. Mean seasonal Chlorophyll-A concentration (mg/m³) and standard error by estuarine region. Concentrations are averaged overall depths, tides and stations within a given estuarine region.



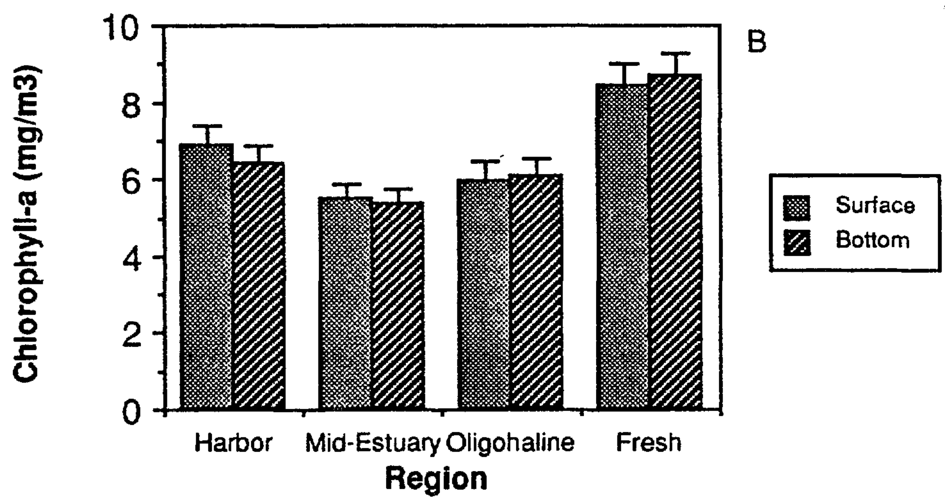


Figure IV.12. Mean Chlorophyll-A concentration (mg/m^3) for surface and bottom for each season (A) or each region (B). Means are averages of tide and either station and month within a season, or station within a region. Error bars are standard errors.

and over all regions of the estuary. This vertical difference indicated a general 33-56% POC enrichment of bottom waters, probably from POC sedimentation from the surface and/or resuspension of bottom organic matter by tidal currents. This vertical pattern correlated with higher oxygen deficits and PO_4 concentrations in the bottom waters (Figure IV.7, IV.23, IV.24) suggesting considerable stimulation of microbial decomposition and remineralization in the bottom waters due to this net input from sedimenting POC.

The vertical gradient in POC was most pronounced at stations in the lower estuarine and upper harbor region (RK 7-12) where bottom water concentrations averaged 2-4 times higher than in the surface waters (Figure IV.10). This pattern suggested that decreasing water velocities in the upper harbor may further enhance POC sedimentation and enrichment of bottom waters.

The gradient of POC along the salinity gradient (Figure IV.10) suggested a general pattern of conservative mixing of POC concentrations from freshwater regions through the estuary. Concentrations in the freshwater and oligohaline regions (1.3-1.5 mg/l) were significantly greater than in mid-estuarine and lower harbor (0.8-1.0 mg/l). The most significant deviation from conservative mixing occurred at the lower estuarine and upper harbor (Stations E and C, salinity range 20-25 ppt, Figure IV.10). Here, surface water concentrations showed a negative deviation (-40%) from conservative mixing, suggesting a net sink of POC from the surface. Furthermore, bottom water concentrations showed a significant positive deviation (+88%) from conservative mixing, suggesting a net source to the bottom water. This pattern further suggests the importance of POC sedimentation in this region of the estuary.

Tributary drainages to the Cooper River estuary were generally similar to adjacent estuarine waters in terms of POC concentrations. The one consistent difference was from the Ashley River where low tide POC concentrations were significantly higher (+0.4 to 0.6 mg/l) than from adjacent harbor stations (A and C, Table IV.7). This input from a relatively developed area may represent an important source of particulate organics to the harbor region, contributing to the apparent increase in harbor POC concentrations (Figure IV.10).

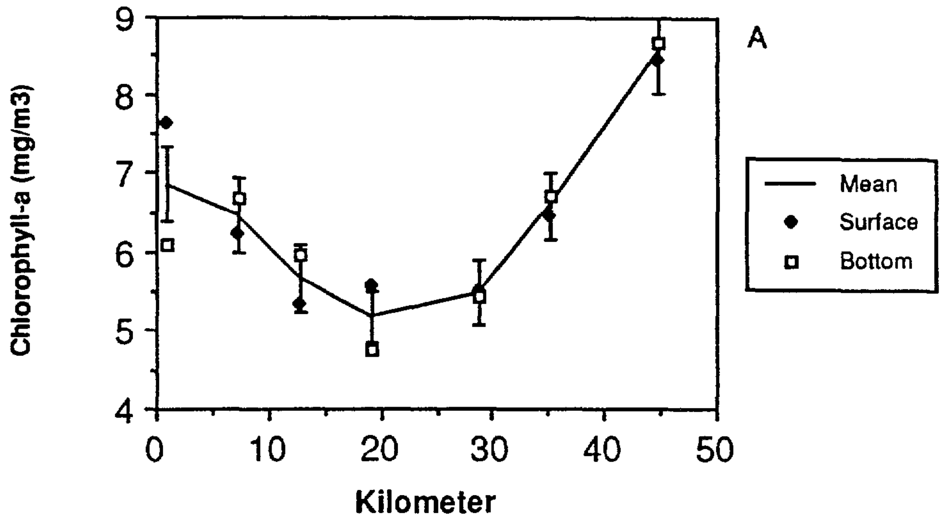
Chlorophyll-a - As an indicator of phytoplankton biomass, chlorophyll-a concentrations varied widely between 1 and 66 mg/m^3, exhibiting significant seasonal and spatial distributions (Appendix IV.A7-IV.A8). Seasonal patterns of chlorophyll concentrations were significantly correlated with turbidity and POC concentrations (Figure IV.11 and IV.12, Appendix IV.B1), with all three parameters exhibiting summer and winter

Table IV.7. Particulate Organic Carbon differences (mg/l) between tributary inputs and main channel. (Low tide concentrations averaged through the water column over the entire study period, n = 21-22, $\bar{X}t$ = tributary mean, $\bar{X}c$ = main channel mean, * indicates significant differences, $p \leq .05$.)

TRIBUTARY	$\bar{X}t$	±SE	MAIN CHANNEL STA	MAIN CHANNEL $\bar{X}c$	MAIN CHANNEL ±SE	Difference ($\bar{X}t$-$\bar{X}c$)
East Branch	1.1	0.1	TW	1.0	0.1	-0.1*
Grove Creek	1.4	0.1	J	1.5	0.1	-0.1*
			L	1.4	0.1	0.0
Goose Creek	1.2	0.1	G	0.9	0.1	0.3*
			H	1.6	0.1	-0.4*
Wando River	0.9	0.1	C	1.2	0.1	-0.3*
			E	0.8	0.1	0.1
Ashley River	1.6	0.1	A	1.0	0.1	0.6*
			C	1.2	0.1	0.4

peaks. The winter peak in chlorophyll was dominated by high concentrations (20-30 mg/m^3) in the fresh and oligohaline regions, while the summer peak was dominated by similar levels in the mid-estuarine and harbor regions. The summer peaks in phytoplankton biomass often dominated the particulate organic matter carbon in the surface waters, typically accounting for 54-75% of the total POC.

Spatial trends in chlorophyll distributions were characterized by chlorophyll concentrations which were lower in the mid-estuarine reaches than at either end of the salinity gradient (Figure IV.13, Appendix IV.A8). Averaged over the entire year, concentrations in the inflowing fresh water (8-9 mg/m^3) were significantly higher than concentrations in the mid-estuarine region (5-6 mg/m^3). Farther downstream, chlorophyll tended to increase again in the harbor region. This pattern resulted in a distinct negative deviation (-33%) from the conservative mixing line, suggesting a net loss of phytoplankton biomass through the estuary. This pattern is distinctly different from those observed in larger embayment-type estuaries such as the Chesapeake Bay and Delaware Bay which often exhibit chlorophyll maxima within the estuary (Fisher *et al.*, 1988; Schemel and Hager, 1986). Such chlorophyll peaks typically occur seaward of the oligohaline turbidity maximum, where clearing estuarine waters allow greater light penetration and stimulation of primary production throughout the water column.



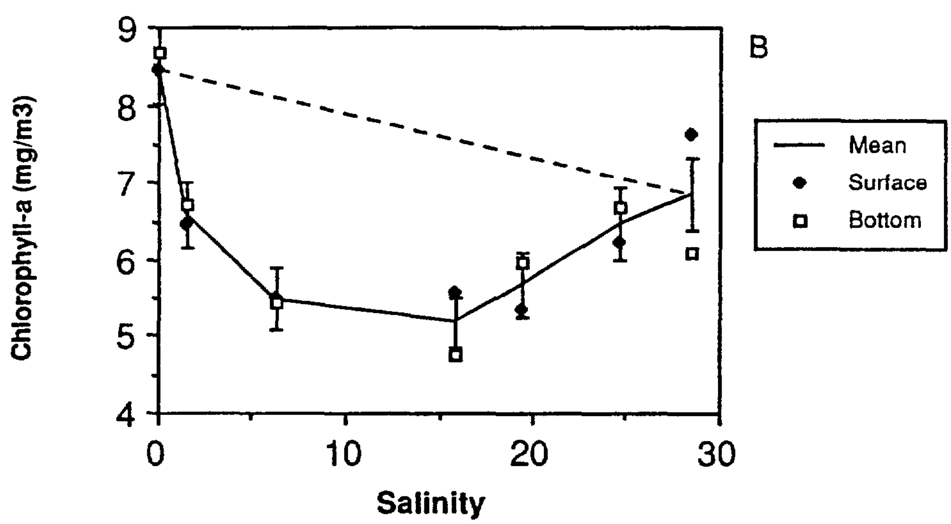


Figure IV.13. Mean Chlorophyll-A concentration (mg/m^3) for the entire sampling period. Means are averages of depth, tide and month. A - Concentration versus river km from the harbor. B - Concentration versus mean salinity for each station along the estuarine gradient. Dashed line represents expected concentration based on conservative mixing.

A major difference between the Cooper River estuary and the other embayment-type estuaries may be related to the residence time of the estuarine water. The residence times of the Chesapeake and Delaware Bays (200 and 80 days, respectively) allow sufficient time for the plankton community to succeed in transition from freshwater and salt water environments, allowing considerable development and production of estuarine phytoplankton. The water residence time in the Cooper River estuary (5-10 days) may be too short for such a transition to take place within the estuary. The partial recovery of chlorophyll concentrations in the harbor area may represent the initial development and production of estuarine and marine phytoplankton.

Inflow from the Ashley River may also have contributed to the increasing chlorophyll levels in the harbor region. While most of the tributary inputs contributed higher chlorophyll concentrations to the main channel, the Ashley River contributions were the highest (Table IV.8). Over the entire year, chlorophyll concentrations at the mouth of the Ashley River averaged over 10 mg/m³ which was 31-52% higher than in the adjacent waters of the harbor. Higher phytoplankton in the Ashley is probably related to more eutrophic conditions, with significantly higher concentrations of NO_3 and PO4 than in adjacent harbor stations (Tables IV.12 and IV.13). The only tributary which had significantly less chlorophyll than the main river channel was the East Branch of the Cooper River (TE, Table IV.8) which drains lowland forested areas of the Francis Marion National Forest.

Table IV.8. Chlorophyll-a differences (mg/m³) between tributary inputs and main channel. (Low tide means averaged through the water column over the entire study period, n = 21-22, $\bar{X}t$ = tributary mean, $\bar{X}c$ = main channel mean, * indicates significant differences, p≤.05.)

TRIBUTARY	$\bar{X}t$	±SE	MAIN CHANNEL STA	MAIN CHANNEL $\bar{X}c$	MAIN CHANNEL ±SE	Difference ($\bar{X}t$-$\bar{X}c$)
East Branch	8.7	1.0	TW	9.9	1.2	-1.2*
Grove Creek	10.6	1.6	J	7.2	1.0	3.4*
			L	9.8	1.5	0.8
Goose Creek	8.3	1.1	G	6.2	1.2	2.1*
			H	7.3	1.2	1.0*
Wando River	8.0	1.2	C	7.5	0.9	0.5
			E	7.4	1.3	0.6
Ashley River	11.4	1.6	A	8.7	1.5	2.7*
			C	7.5	0.9	3.9*

Dustan and Pickney (1989) recently reported on chlorophyll patterns in Charleston Harbor, postulating that tidally-induced aggregations at frontal zones may represent an important mechanism controlling phytoplankton distributions and planktonic trophic dynamics in this system. Our highest chlorophyll observation (66 mg/m^3) was from the western side of the harbor (station B) in an area often characterized by frontal discontinuities between the Ashley River inflow and the main water mass of the harbor.

Dissolved Organic Carbon - DOC concentrations were quite variable ranging from 0.2 to 15.5 mg/l with an overall mean of 3.5 ± 0.2 mg/l over the main axis of the Cooper River estuary. Typical of most natural waters, DOC constituted most of the total organic carbon in the water column (62-84%, averaged over all seasons and regions).

Although there were no distinct seasonal patterns, there was a significant increase in concentrations between summer and fall (Figure IV.14 and IV.15, Appendix IV.A10). The most significant monthly change in DOC concentrations was a 3 to 4-fold increase between August and late September which corresponded to the transition between low summer flows and higher autumn flows in the Cooper River. After a sustained period of lower flows (July through early September, Figure IV.2), the autumn increase in river discharge may have mobilized accumulated decay products in contributing watersheds and wetlands. This mechanism is also consistent with a significant correlation between DOC and river discharge (Appendix IV.B1), which may have been more pronounced after the summer period of low flows. Similar patterns have been documented in large floodplain rivers where floodplain inundation, particularly after prolonged periods of low flow, results in substantially higher levels of organic transport in the river (Elder and Mattraw,1982; Osemene, 1985). This effect may represent an important input of highly labile substrates for rapid assimilation into aquatic food webs during the fall when many organisms are beginning to utilize estuarine areas.

DOC also varied significantly with depth in the water column and with distance along the salinity gradient (Appendix IV.A9). Vertical patterns exhibited consistently higher concentrations in the surface waters (by 15-20%), particularly in the fall and in the harbor region, where surface water concentrations averaged 50-60% higher than in bottom waters (Figure IV.15 and IV.16). The higher concentrations in the fall were probably related to the autumn increase in river flow (as discussed above), with more pronounced effects in the less saline surface waters.

The higher concentrations of DOC in the surface waters of the harbor region was most evident at the mouth of the harbor, where surface concentrations averaged almost

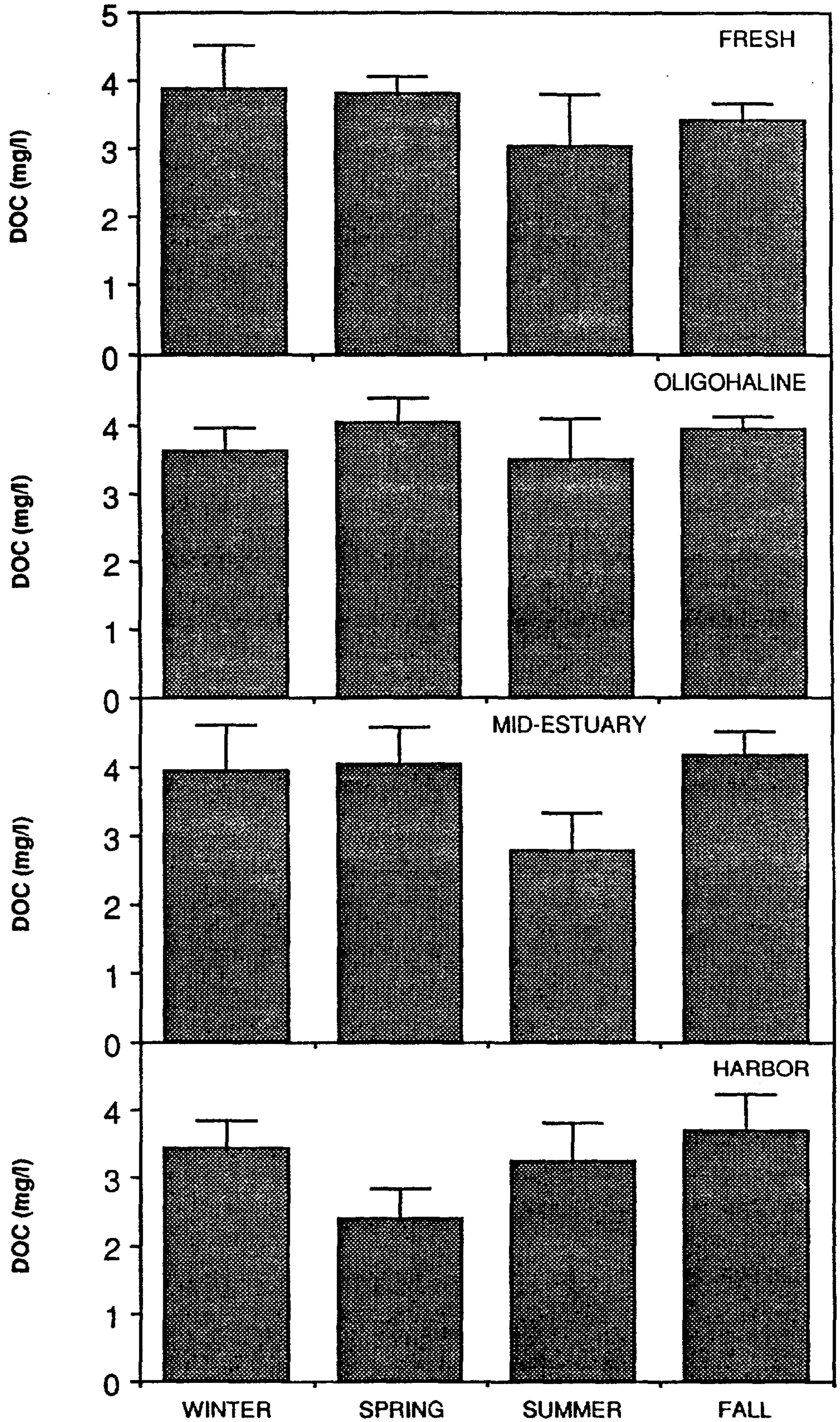
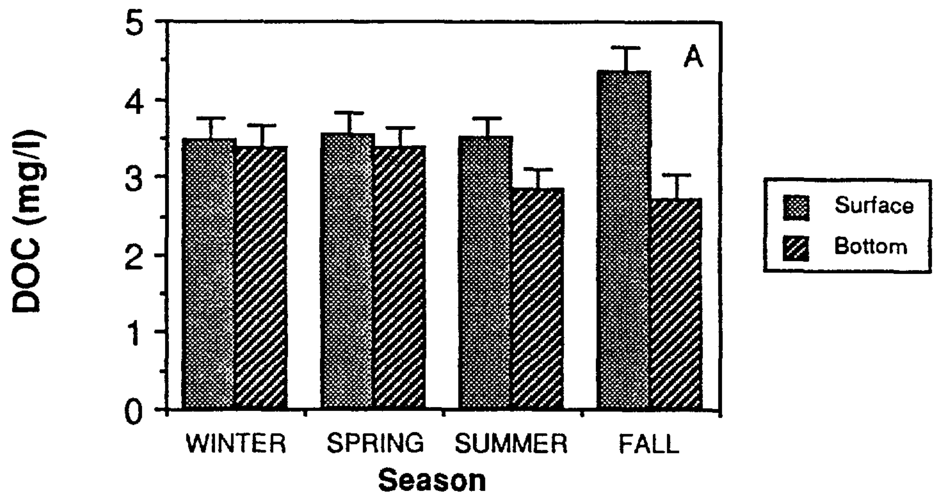


Figure IV.14. Mean seasonal dissolved organic carbon (DOC) concentration (mg/l) and standard error by estuarine region. Concentrations averaged overall depths, tides and stations within a given estuarine region.



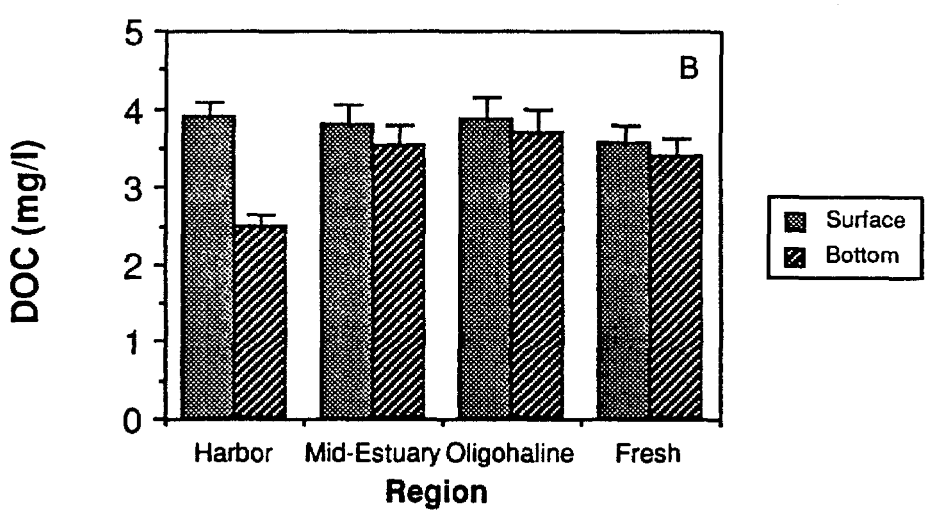
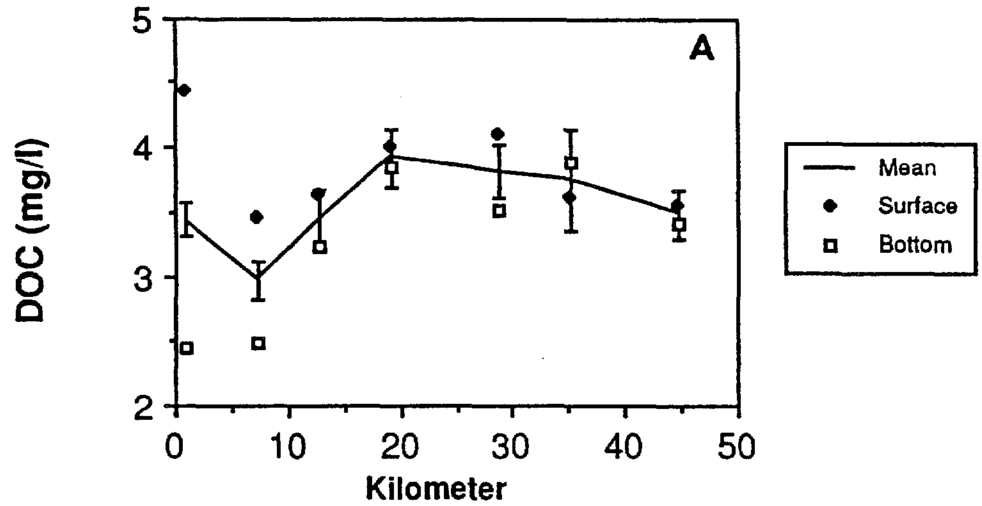


Figure IV.15. Mean dissolved organic carbon concentration (mg/l) for surface and bottom for each season (A) or each region (B). Means are averages of tide and either station and month within a season (A) or season and station within a region (B). Error bars are standard errors.



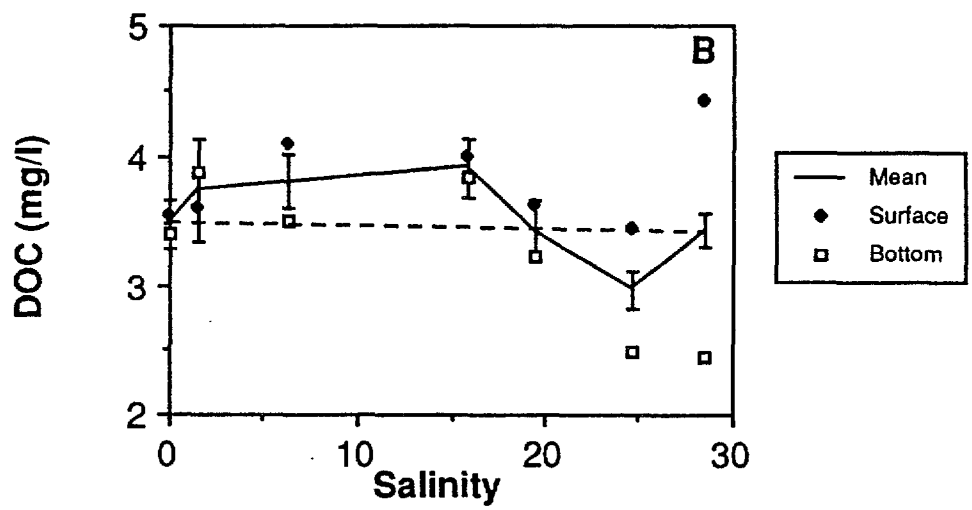


Figure IV.16. Mean dissolved organic carbon concentration (mg/l) for the entire sampling period. Means are averages of depth, tide and month. (A) Concentration versus river km from the harbor. (B) Concentration versus mean salinity for each station along the estuarine gradient. Dashed line represents expected concentration based on conservative mixing.

2-fold higher than in bottom waters (Figure IV.16). This pattern suggests a source of DOC in the surface waters at the harbor or ocean end of the salinity gradient. Potential DOC sources to the harbor could have been the Ashley River (which had 55 to 100% higher DOC concentrations than adjacent harbor stations (Table IV.9)), as well as the CCPW wastewater treatment plant (which is permitted to discharge 18 million gallons per day (MGD) to the harbor along the southwestern shore (Figure IV.1, Table IV.1)).

Table IV.9. Dissolved organic carbon differences (mg/l) between tributary inputs and main channel. (Low tide means averaged through the water column over the entire study period, n = 21-22, $\bar{X}t$ = tributary mean, $\bar{X}c$ = main channel mean, * indicates significant differnces, p≤.05.)

			MAIN CHANNEL			
TRIBUTARY	$\bar{X}t$	±SE	STA	$\bar{X}c$	±SE	Difference ($\bar{X}t$-$\bar{X}c$)
East Branch	4.2	0.5	TW	3.0	0.4	1.3*
Grove Creek	4.5	0.5	J	3.7	0.5	0.8*
			L	3.7	0.5	0.8*
Goose Creek	4.0	0.7	G	3.8	0.4	0.2
			H	3.8	0.6	0.2
Wando River	4.0	0.4	C	2.5	0.4	1.5*
			E	3.4	0.6	0.6
Ashley River	5.0	0.6	A	3.2	0.4	1.8*
			C	2.5	0.4	2.5*

The spatial distribution of DOC across the estuarine salinity gradient exhibited distinctly different patterns for surface and bottom waters (Figure IV.16). The peak near the center of the salinity gradient was apparently due to elevated bottom water concentrations. This pattern could be due to subsurface discharges from the Westvaco paper mill, which discharges up to 30 MGD of organic-rich effluent at this point in the estuary. This is the largest source of organic wastewater in the Cooper River (Table IV.1) and could dominate organic matter concentrations in the areas adjacent to the discharge. However, the effect was only apparent in the bottom waters over an 8-10 RK distance downstream from the paper mill.

Inorganic Nutrients:

Two inorganic dissolved nitrogen species (ammonium and nitrate-nitrite) and one dissolved phosphorus species (ortho-phosphate) were investigated to determine spatial and temporal variation along the salinity gradient of the Charleston Harbor-Cooper River estuary. Inorganic nutrients reflect internal processes as well as external point and nonpoint sources of nutrient discharge.

Ammonium - NH4 varied widely during the study period, ranging from nondetectable concentrations (ND) to 79.3 ug-at/l, and with a mean concentration of 5.24 ug-at/l ($\pm$ 0.32). No significant differences were observed for tidal stage or depth (Table IV.10, Appendix IV.A11). However, ammonium exhibited significant differences in spatial and seasonal distributions (Figures IV.17 and IV.18, Appendix IV.A11-IV.A12). The freshwater region had the lowest ammonium concentration (1.4 $\pm$ 0.24 ug-at/l) with mid-estuarine stations exhibiting the highest concentrations within the Cooper River (Figure IV.18).

Table IV.10. Ammonia concentration (ug-at N/L) averaged over the seven main channel stations for the entire period by tide, depth and month.

		EXTREMES	MEAN	STANDARD ERROR	NUMBER
TIDE:	High	0.0-79.3	5.06	.47	150
	Low	0.1-74.6	5.4	.47	166
DEPTH:	Surface	0.0-74.6	5.51	.48	158
	Bottom	0.0-79.3	4.98	.46	158
MONTH:	January	0.0-20.7	3.0	.8	28
	February	0.2-30.8	3.1	1.5	12
	March	0.4-13.6	4.6	1.1	26
	April	0.8-17.5	7.3	1.1	28
	May	1.1-17.6	6.1	1.1	28
	*June	0.2-12.8	5.1	1.0	28
	July	0.6-74.6	11.1	2.7	28
	*August	0.1-12.4	1.9	.4	28
	September	0.8-49.4	3.9	.8	28
	October	1.4-64.1	6.3	.9	26
	November	1.4-17.9	7.3	1.2	28
	December	0.6-79.3	7.1	1.6	28

* Spring tide only

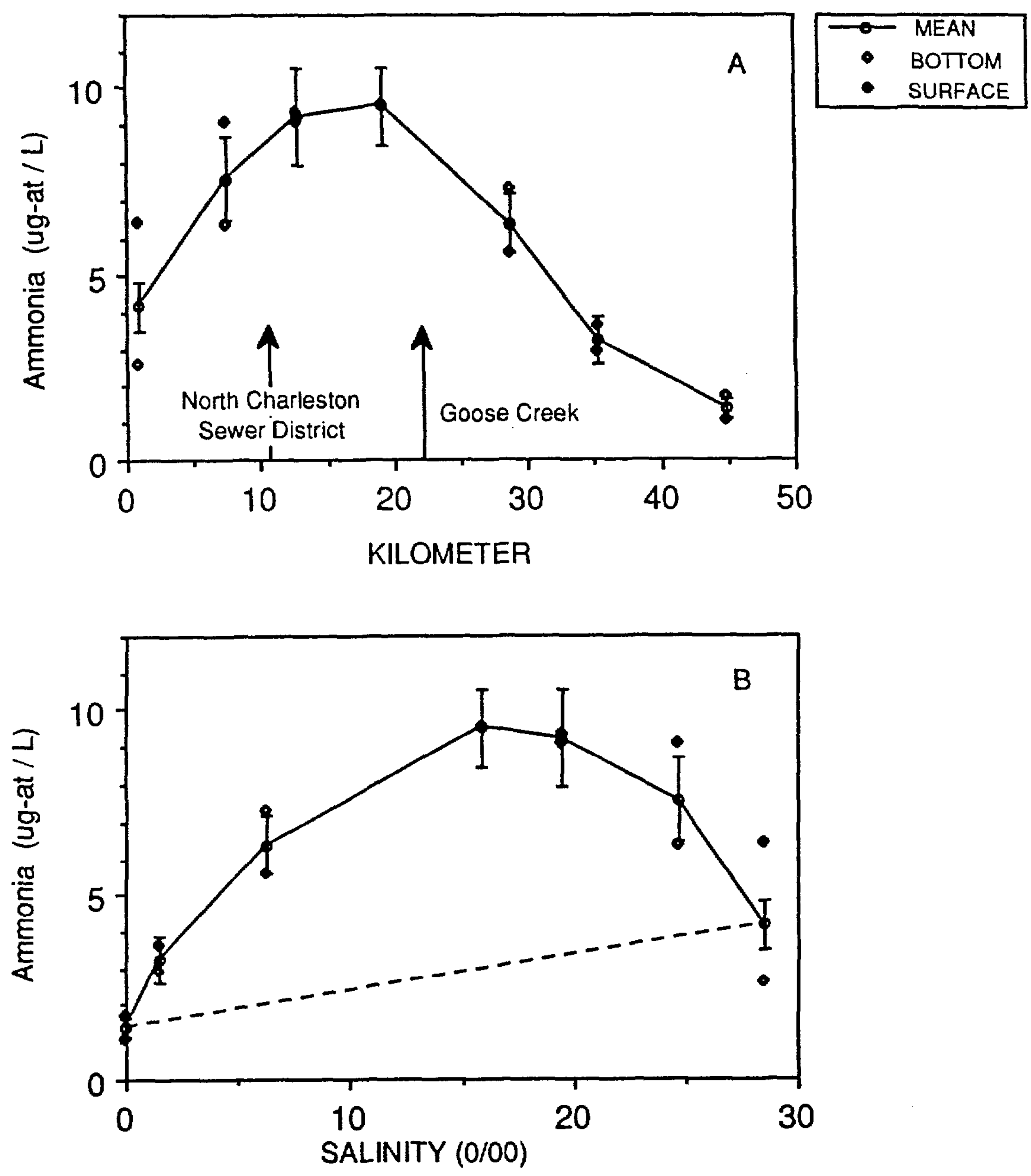
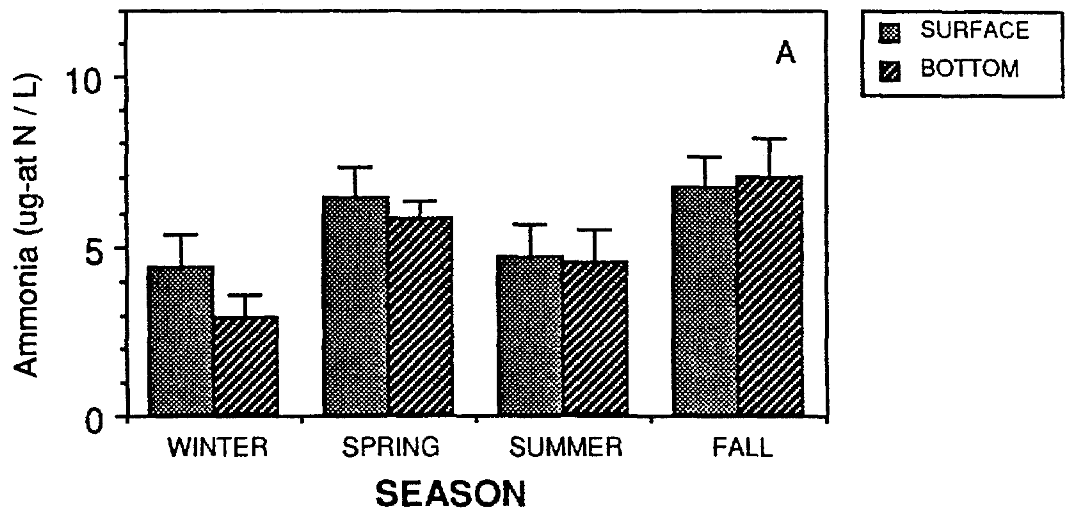


Figure IV.17. Mean ammonia concentration (ug-at N/L) for the entire sampling period. Means are averages of depth, tide and month. (A) Concentration versus river km from the harbor. (B) Concentration versus mean salinity for each station along the estuarine gradient. Dashed line represents expected concentration based on conservative mixing.



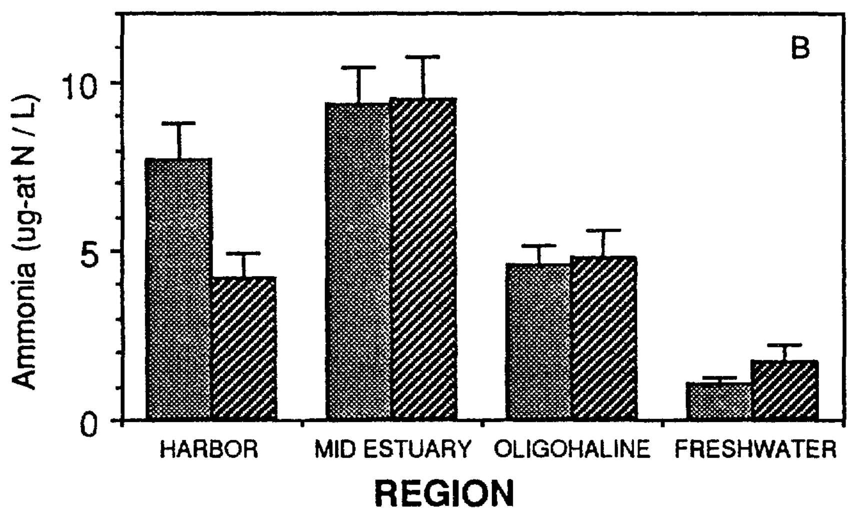


Figure IV.18. Mean ammonia concentration (ug-at N/L) for surface and bottom for each season (A) or each region (B). Means are averages of tide and either station and month within a season (A) or season and station within a region (B). Error bars are standard errors.

Although there were significant average monthly differences, no obvious monthly patterns were observed. Both the highest (July - 11.08 + 2.71 ug-at/l) and the lowest (August - 1.9 + 0.43 ug-at/l) mean monthly concentrations occurred during the summer months (Table IV.10). Fall ammonium concentrations were significantly higher than summer and winter concentrations (Figure IV.18, Appendix IV.A12). Winter concentrations were significantly lower than spring and fall concentrations. Differences with depth were only observed during winter, with surface concentrations greater than bottom concentrations suggesting that, in general, ammonia is relatively well mixed in the Cooper River.

Differing processes were important in regulating ammonium dynamics in the different estuarine regions. In the freshwater region, bottom concentrations (1.73 + 0.48 ug-at/l) were significantly higher than surface concentrations (1.08 + .22 ug-at/l) (Figure IV.19). Higher concentrations in the bottom waters of the freshwater region suggest that internal recycling may be supplying the ammonium detected (Gilbert, 1982 in Fisher *et al.*, 1988). In the freshwater region, concentrations were lowest in the winter and highest and most variable during the fall (8 times higher than winter). Processes internal to the estuarine system (e.g. decomposition of wetland vegetation and concurrent marsh runoff or benthic fluxes) may be responsible for the fall peak in ammonium.

Several other investigators have noted higher ammonium concentrations during the fall months in estuarine waters and higher fluxes of ammonium from marshes during this season (Stevenson *et al.*, 1977; Whiting *et al.*, 1988). Annual average benthic ammonium fluxes in coastal sediments range from 68 to 295 ug-at/m2/hr, and the seasonal variation in the flux rate is related to temperature and organic matter inputs. These factors contribute to higher fluxes during late summer and early fall (Boynton *et al.*, 1980).

The ammonium distribution was very non-conservative in the Cooper River and Charleston Harbor basin. Measured concentrations suggest that both external inputs and internal recycling regulate the ammonium within the mid-estuary. The salinity- nutrient mixing diagram indicated a significant point source within the mid-estuary region (Figure IV.17) with elevated concentrations both in the surface and bottom water (9.30 + 1.13 and 9.45 + 1.25 ug-at/l, respectively). Maximum ammonium concentrations were measured between 16 to 18 ppt salinity with lower concentrations at both the freshwater and high salinity stations. Oceanic waters and Cooper River riverine input are, therefore, not the major ammonium sources. Ammonium concentrations in the mid-estuary region were over 200% higher than would be predicted from conservative mixing. Tributary inputs within the mid-estuary region were not responsible for the measured ammonium concentrations. No tributary input in the mid-estuarine region was significantly different from adjacent stations in the main channel of the estuary (Table IV.11).

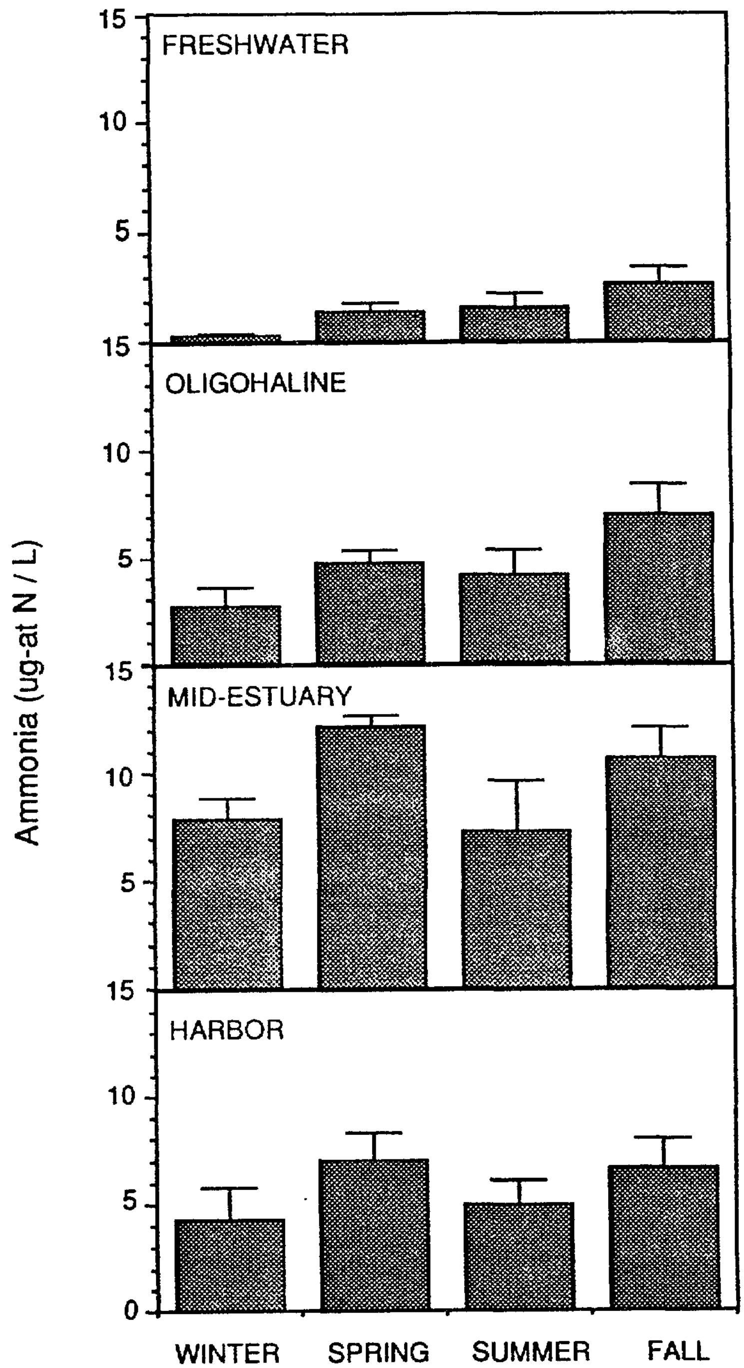


Figure IV.19. Mean seasonal ammonia concentrations (ug-at N/L) and standard error by estuarine region. Means are averages of tides, depths and stations within a given estuarine region.

Table IV.11. Ammonium differences (ug-at N/L) between tributary inputs and the main channel. (Low tide means averaged through the water column over the entire study period, n = 22-24, $\bar{X}t$ = tributary mean, $\bar{X}c$ = main channel mean, * indicates significant differences, p<.05)

			MAIN CHANNEL			
TRIBUTARY	$\bar{X}t$	$\pm$SE	STA	$\bar{X}c$	$\pm$SE	Difference ($\bar{X}t$-$\bar{X}c$)
East Branch	2.0	0.7	TW	1.8	0.4	0.2
Grove Creek	2.4	0.6	J	4.9	0.6	2.5
			L	2.5	0.7	0.1
Goose Creek	6.8	1.1	G	8.3	1.4	1.5
			H	7.0	1.3	0.2
Wando River	7.0	1.2	C	11.3	1.8	4.3*
			E	9.6	2.2	2.6
Ashley River	7.0	1.2	A	7.1	1.3	0.1
			C	11.3	1.8	4.3*

There were two potential point sources of ammonium discharge in the mid-estuarine region (Table IV.1). A non-bleach Kraft paper mill (Westvaco, located at Station G) and a municipal waste water treatment plant (NCSD, located at river km 10.8, between stations C and E). Although no ammonium discharge data were available for Westvaco, the NCSD has the highest permitted ammonium discharge for the river (Table IV.1) with an average 8,661 kg ammonium per day during the study period (SCDHEC 1989). While the location of the maximum ammonium concentration would suggest the Westvaco discharge, calculated maximum ammonium deviation from conservative mixing (252% at 12 ppt) suggests that the NCSD wastewater treatment plant was the primary source. Our sampling design may not have been adequate to locate the true position of the concentration maximum.

If in fact the NCSD was the source for the elevated ammonium at Stations E and G, the data suggested a significant upstream transport with higher salinity bottom waters. Detailed analyses of the data substantiated this observation (Table IV.11). Low tide ammonium concentrations were higher at Station C (11.30 ug-at/l) than stations E (9.57 ug-at/l) or G (8.27 ug-at/l). Also, when the spatial variation of high and low tide ammonium concentrations were analyzed during a given month, the potential contribution from the NCSD was supported. From March to June, maximum ammonium concentrations were detected at Station G at high tide with a steep gradient of increasing concentration occurring from Station C to G in the bottom water samples. At low tide, the highest concentrations occurred at Station C again with a steep gradient of increasing concentration from Station G, E to C. These data suggest an upstream transport of ammonium from a point source with flooding waters and a down stream transport from the point source with ebbing waters. The 12 ppt area was substantially upstream from the NCSD discharge location (at ~22 ppt). The low freshwater discharge occurring in the Cooper River and large tidal influence may have contributed to the observed significant upstream transport of nutrients.

Seasonal data support this conclusion and suggest that internal processes may be adding to the upstream transport during the fall. In the mid-estuary region, ammonium concentrations were highest in the spring coincident with maximum ammonium discharge from the NCSD waste water treatment plant (averaging 10,407 kg/day). A second peak in ammonium occurred during the fall. Lower discharges of ammonium from the NCSD waste water treatment plant during the fall (5949 kg/day) indicate that the fall peak in ammonium in the mid-estuary region may have reflected internal processing of the organic matter accumulated during the spring and summer months and concurrent benthic ammonium releases.

Concentrations of ammonium within the harbor region were a function of both point and non-point sources. The Ashley River appeared to be an important source of ammonium for the harbor with significantly higher concentrations in the harbor surface waters (7.70 + 1.09 ug-at/l) than bottom waters (4.21+ 0.69 ug-at/l). Within the harbor region Station B was significantly higher than Station A and C. The Plum Island Sewage Treatment Plant (CCPW, Figure IV.1) was located near Station B and discharged an average of 105 kg ammonium per day during the study period (Fairey, pers. comm.). Higher concentrations in the surface waters of the harbor indicated that internal recycling is minor relative to external inputs from the Cooper River and Plum Island point source.

The elevated concentrations and lack of difference in surface and bottom waters in the oligohaline region suggest an external source and a high degree of mixing. The oligohaline region appears to be influenced by both the freshwater and mid-estuary regions. Oligohaline region ammonium concentrations were intermediate between mid-estuary and freshwater regions, with no differences with depth. Seasonal patterns are similar to those observed in the freshwater region but at concentrations intermediate between the two regions. The harbor region seasonal patterns are similar to the mid-estuary seasonal patterns but at lower levels, indicating a dilution of the ammonium contributions from the mid-estuary.

Nitrate-Nitrite - NO_3-NO_2 concentrations averaged 3.65 ± 0.18 ug-at/l and ranged from ND to 24.5 ug-at/l. No significant tidal differences were noted, with concentrations averaging 3.58 ± 0.25 ug-at/l at high tide and 3.72 ± 0.24 ug-at/l at low tide (Table IV.12, Appendix IV.A13). Nitrate-nitrite exhibited significant depth and season distributions (Appendix IV.A13-IV.A14). Concentrations in bottom waters (4.41 ± 0.24 ug-at/l) were higher than those for surface waters (3.90 ± 0.26 ug-at/l). Late summer months (August - 7.45 ± 1.83 ug-at/l, September - 7.22 ± 1.68 ug-at/l) had the higher nitrate-nitrite concentrations and greater variability while winter and fall months (ie. November - 2.34 ± 0.17, December - 2.7 ± 0.25 ug-at/l) had significantly lower nitrate-nitrite (Figure IV.20) and lower variability. Concentrations were significantly higher in summer than during all other seasonal periods while winter, spring, and fall were not significantly different from each other. The lowest nitrate-nitrite concentration occurred in March (2.19 ± 0.24 ug-at/l).

Table IV.12. Nitrate-nitrite concentrations (ug-at N/L) averaged over the seven main tributary stations for the entire sampling period by tide, depth and month.

		EXTREMES	MEAN	STANDARD ERROR	NUMBER
TIDE:	High	0.3-21.9	3.6	0.3	150
	Low	0.1-24.5	3.7	0.2	166
DEPTH:	Surface	0.1-22.5	3.9	0.3	158
	Bottom	0.1-24.5	3.4	0.2	158
MONTH:	January	0.9-4.3	2.6	0.2	28
	February	1.3-11.6	5.4	1.3	12
	March	0.5-4.2	2.2	0.2	26
	April	0.3-7.1	3.7	0.4	28
	May	0.7-7.1	4.0	0.4	28
	*June	0.1-5.9	2.9	0.4	28
	July	0.3-6.7	3.0	0.5	28
	*August	0.2-2.0	7.5	1.8	28
	September	0.8-24.5	7.2	1.7	28
	October	0.6-11.9	4.1	.7	26
	November	0.7-4.2	2.3	.2	28
	December	0.6-4.3	2.7	.3	28

* Spring tide only

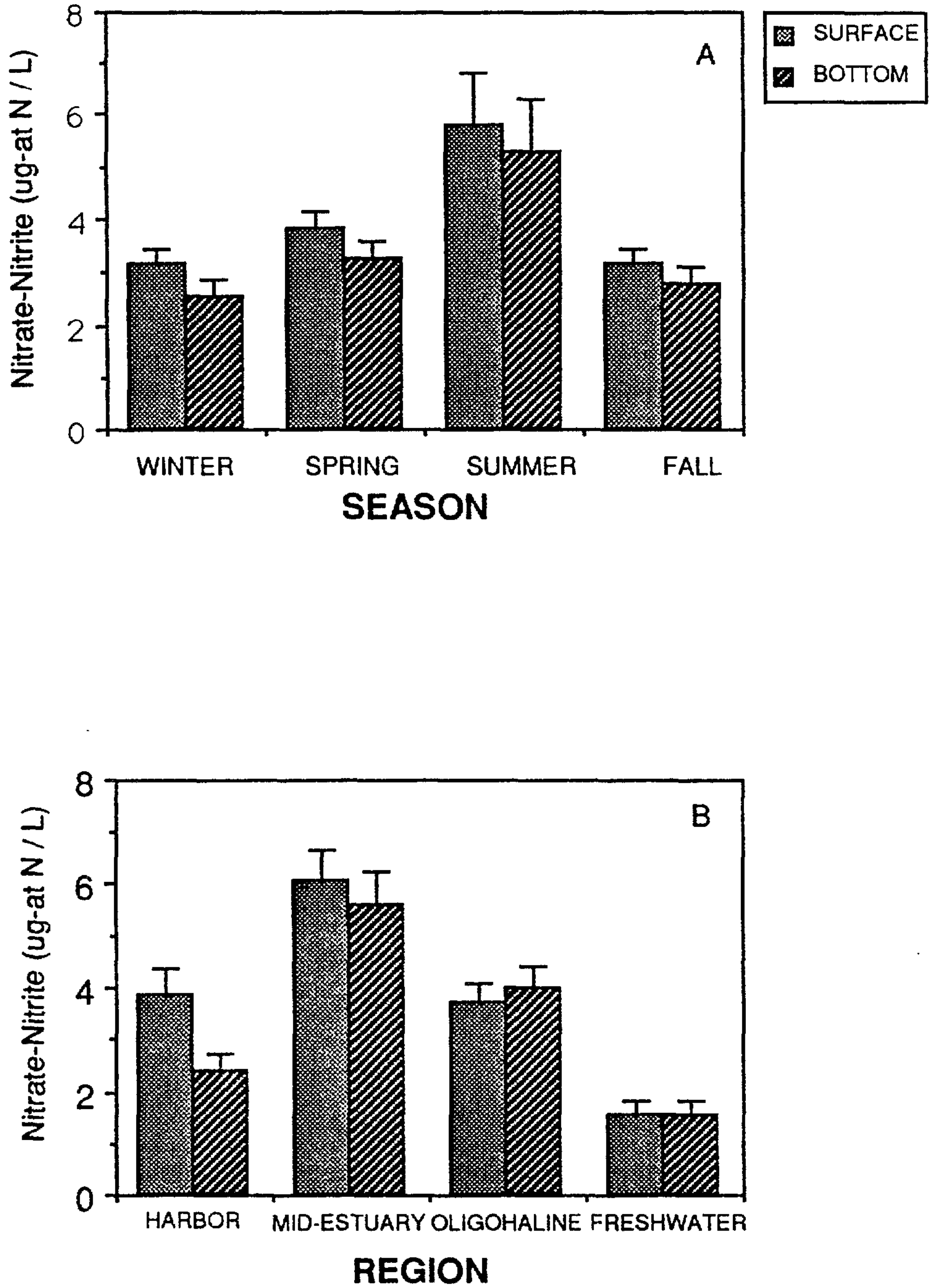
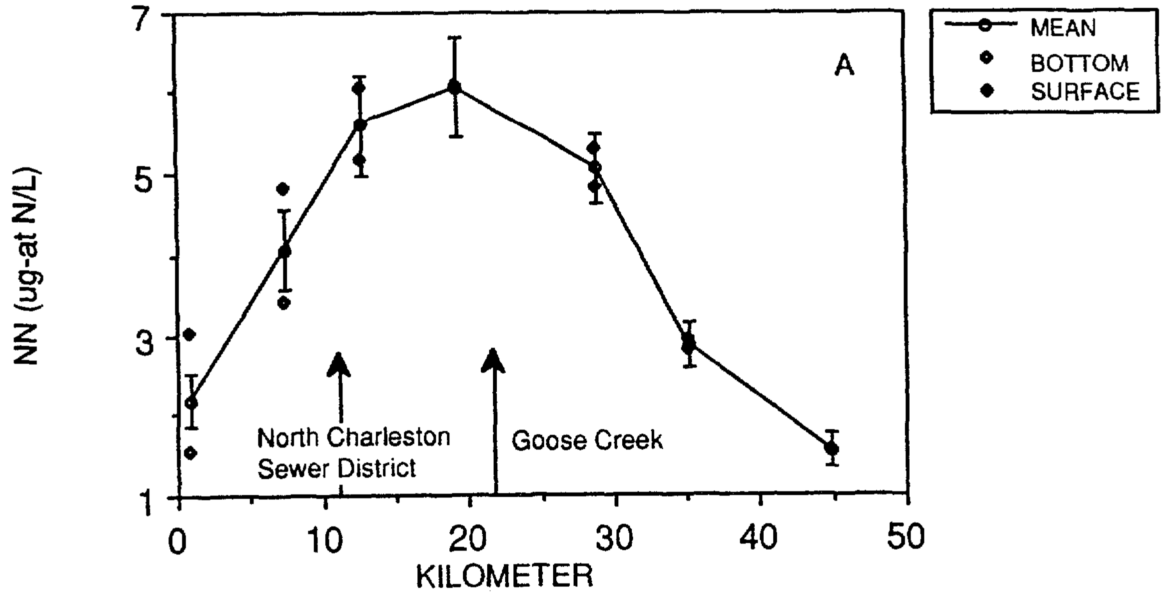


Figure IV.20. Mean nitrate-nitrite concentration (ug-at N/L) for surface and bottom for each season (A) or each region (B). Means are averages of tide and either station and month within a season (A) or season and station within a region (B). Error bars are standard errors.

Significant differences occurred in the nitrate-nitrite concentrations between stations (Figure IV.21, Appendix IV.A13-IV.A14). Station O (freshwater) had the lowest nitrate-nitrite concentration (1.55 $\pm$ 0.21 ug-at/l) whereas a mid-estuary station (Station G) had the highest nitrate-nitrite concentration (6.06 $\pm$ 0.63 ug-at/l). The mid-estuary stations (stations E, G, and J) and harbor Station C were not significantly different from each other. The remaining stations A, L, and O were not significantly different from each other but were significantly different from the mid-estuary stations. When the stations were grouped into regions based on salinity, each region was significantly different from each other with the lowest concentrations in the freshwater region (1.55 $\pm$ 0.30 ug-at/l) and highest in the mid-estuary region (6.07 $\pm$ 0.61 - surface, 5.59 $\pm$ 0.65 - bottom; ug-at/l) (Figure IV.20, Appendix IV.A14). A significant interaction between station and depth was identified; however, only in the harbor region where surface concentrations (3.88 $\pm$ 0.50 ug-at/l) were significantly higher than bottom concentrations (2.39 $\pm$ 0.34 ug-at/l), suggesting an external source of nitrate-nitrite. In the freshwater, oligohaline and mid-estuary regions, high tide concentrations were higher than low tide concentrations, while in the harbor the low tide concentrations were higher than high tide.

Seasonal patterns varied with region in the estuary (Figure IV.22). The lowest concentration (0.57 $\pm$ 0.10 ug-at/l) occurred during the summer in the freshwater region while the highest concentration (13.3 $\pm$ 1.68 ug-at/l) occurred during the summer in the mid-estuary region. Summer concentrations in the harbor and oligohaline regions were intermediate to those in the freshwater and mid-estuary regions. In the freshwater region highest concentrations occurred during the winter season while the lowest concentration occurred during the same season in the mid-estuary region. No seasonal patterns were obvious in the oligohaline region. Seasonal patterns in the harbor region were similar to the mid-estuary region, but at a fraction of the concentration. During the winter, harbor concentrations were 56% of the mid-estuary and, in the summer, 50% of mid-estuary concentrations.

Two analyses indicate a significant source of nitrate-nitrite within the harbor between stations A and C, and within the mid-estuary region near stations E and G. Within the harbor region both Station B and the Ashley River nitrate-nitrite were significantly higher than Station A, suggesting potentially both point source (CCPW) and non-point sources (Ashley River) for the harbor. When nitrate-nitrite concentrations at low tide are regressed with salinity for stations A, B and the Ashley River, it appears that the concentrations measured at stations A and B are primarily influenced by discharge from the Ashley River. The nitrate-nitrite concentration declined linearly (R^2 = 0.995) with increased salinity.



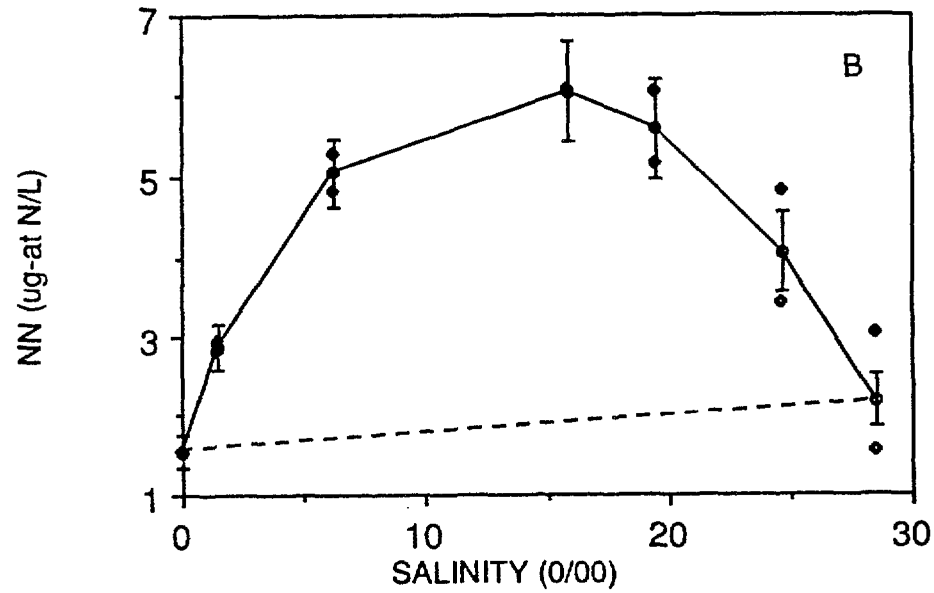


Figure IV.21. Mean nitrate-nitrite concentration (ug-at N/L) for the entire sampling period. Means are averages of depth, tide and month. (A) Concentration versus river km from the harbor. (B) Concentration versus mean salinity for each station along the estuarine gradient. Dashed line represents expected concentration based on conservative mixing.

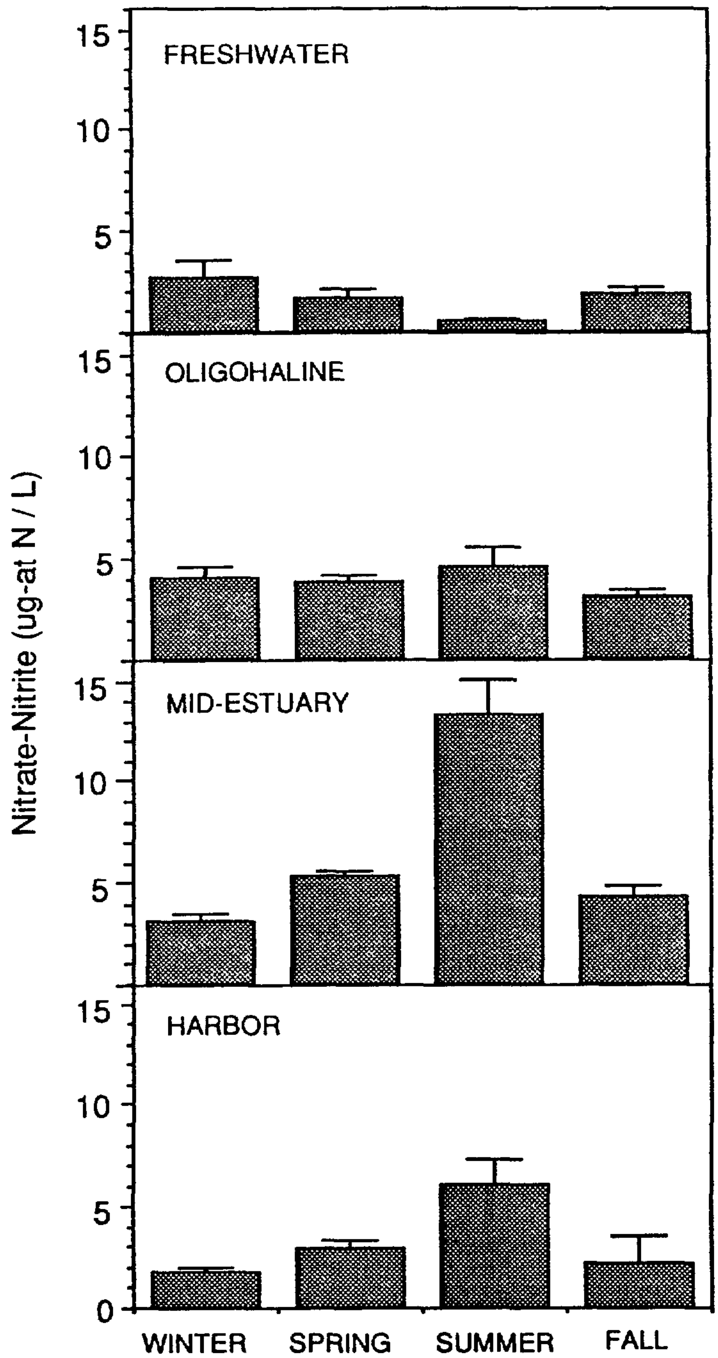


Figure IV.22. Mean seasonal nitrate-nitrite concentrations (ug-at N/L) and standard error by estuarine region. Means are averages of tide, depth and stations within a given region.

The salinity/nutrient diagrams for nitrate-nitrite suggest a source at mid-salinities (16 ppt) with lower concentrations at both freshwater and high salinity stations (Figure IV.21). Oceanic waters and Cooper River riverine input are, therefore, not the nitrate-nitrite sources in the freshwater, oligohaline and mid-estuary portions of the gradient. No tributary inputs within this region were significantly different from adjacent estuarine stations (Table IV.13). The high concentrations in the mid-estuary at stations E and G suggest a linkage between the ammonium discharged and nitrate-nitrite concentrations. Ammonium is rapidly oxidized to nitrate-nitrite in surface waters. Nitrate-nitrite concentrations are maximal one station above maximal ammonium concentrations in flooding waters and one station lower in ebbing waters. The one-station displacement suggests oxidation of the ammonium. Oxygen concentrations exhibited a sag in this region, with the lowest concentrations occurring at Station C.

Table IV.13. Nitrate-nitrite differences (ug-at N/L) between tributary inputs and the main channel. (Low tide concentrations averaged through the water column over the entire study period, n = 22-24, $\bar{X}t$ = tributary mean, $\bar{X}c$ = main channel mean, * indicates significant differences, p<.05.)

			MAIN CHANNEL			
TRIBUTARY	$\bar{X}t$	±SE	STA	$\bar{X}c$	±SE	Difference ($\bar{X}t$-$\bar{X}c$)
East Branch	1.2	0.3	TW	1.6	0.4	0.4
Grove Creek	2.0	0.2	J	4.4	0.5	2.4
			L	2.2	0.4	0.2
Goose Creek	6.4	0.8	G	5.8	0.8	0.6
			H	5.8	0.7	0.6
Wando River	4.3	0.6	C	4.9	0.8	0.6
			E	5.6	0.9	1.3
Ashley River	5.8	0.8	A	3.6	0.6	2.2*
			C	4.9	0.8	0.9

Nitrate-nitrite distribution was very non-conservative in the Cooper River and Charleston Harbor basin. The salinity-nutrient mixing diagram indicated a significant point source within the mid-estuary region (Figure IV.21). Maximum input of nitrate-nitrite occurred between 12 to 14 ppt salinity, somewhat further upstream than the ammonium maximum input. Nitrate-nitrite additions in the mid-estuary region were over 200% higher than would be predicted from conservative mixing. The maximum addition was 244% at 12 ppt. The 12 ppt area is substantially upstream from the NCSD discharge location (at ~22 ppt) but in the same location as the maximum ammonium concentration. Displacement of the maximum nitrate-nitrite input upstream from the ammonium maximum input (16 to 18 ppt) suggested oxidation of ammonium to nitrate-nitrite as it is transported upstream.

Ortho-phosphate - PO_4 occurred in the lowest concentrations of the three nutrient fractions measured. Mean PO_4 concentration was 0.68 $\pm$ 0.04 ug-at/l with a range of ND to 5.20 ug-at/l. Concentrations were higher at high tide (0.81 $\pm$ 0.06, ND-5.2 ug-at/l) than low tide (0.57 $\pm$ 0.05, ND-2.7 ug-at/l) and greater near the bottom (0.79 $\pm$ 0.05, ND-5.2 ug-at/l) than surface (0.58 $\pm$ 0.05, ND-5.2 ug-at/l) (Table IV.14, Appendix IV.A15). The highest PO_4 concentrations occurred in the harbor (Station A,1.16 $\pm$ 0.13 ug-at/l) and the lowest in the freshwater region (Station O, 0.17 $\pm$ 0.05 ug-at/l) (Figure IV.23). Concentrations decreased almost linearly from the harbor to the freshwater region which suggested a harbor source for PO_4. No differences were detected between surface and bottom in the harbor (1.08 vs 1.10 ug-at/l), but for all other regions bottom concentrations were significantly higher (Figure IV.24). The largest difference between surface and bottom occurred in the mid-estuary region (surface 0.71 ug-at/l, bottom 1.12 ug-at/l). Bottom concentrations in the mid-estuary were equivalent to concentrations in the harbor region, accounting for the lack of statistical difference between the two regions and suggesting upstream transport of PO_4 with higher density waters. Both the freshwater and oligohaline regions were significantly different from each other and from the harbor and mid-estuary regions.

Distinct seasonal patterns in PO_4 concentrations occurred with spring and fall having similar concentrations and no significant difference between surface and bottom (Figure IV.24, Appendix IV.A16). Summer PO_4 concentrations were significantly higher than those in winter with concentrations in bottom waters (0.99 ug-at/l - bottom, 0.62 ug-at/l - surface) greater than surface waters. Monthly concentrations were significantly different with the highest concentrations occurring in July (0.99 $\pm$ 0.19, ND-2.70 ug-at/l) and lowest concentrations occurring in February (0.23 $\pm$ 0.07, 0.03-0.99 ug-at/l) (Table IV.14).

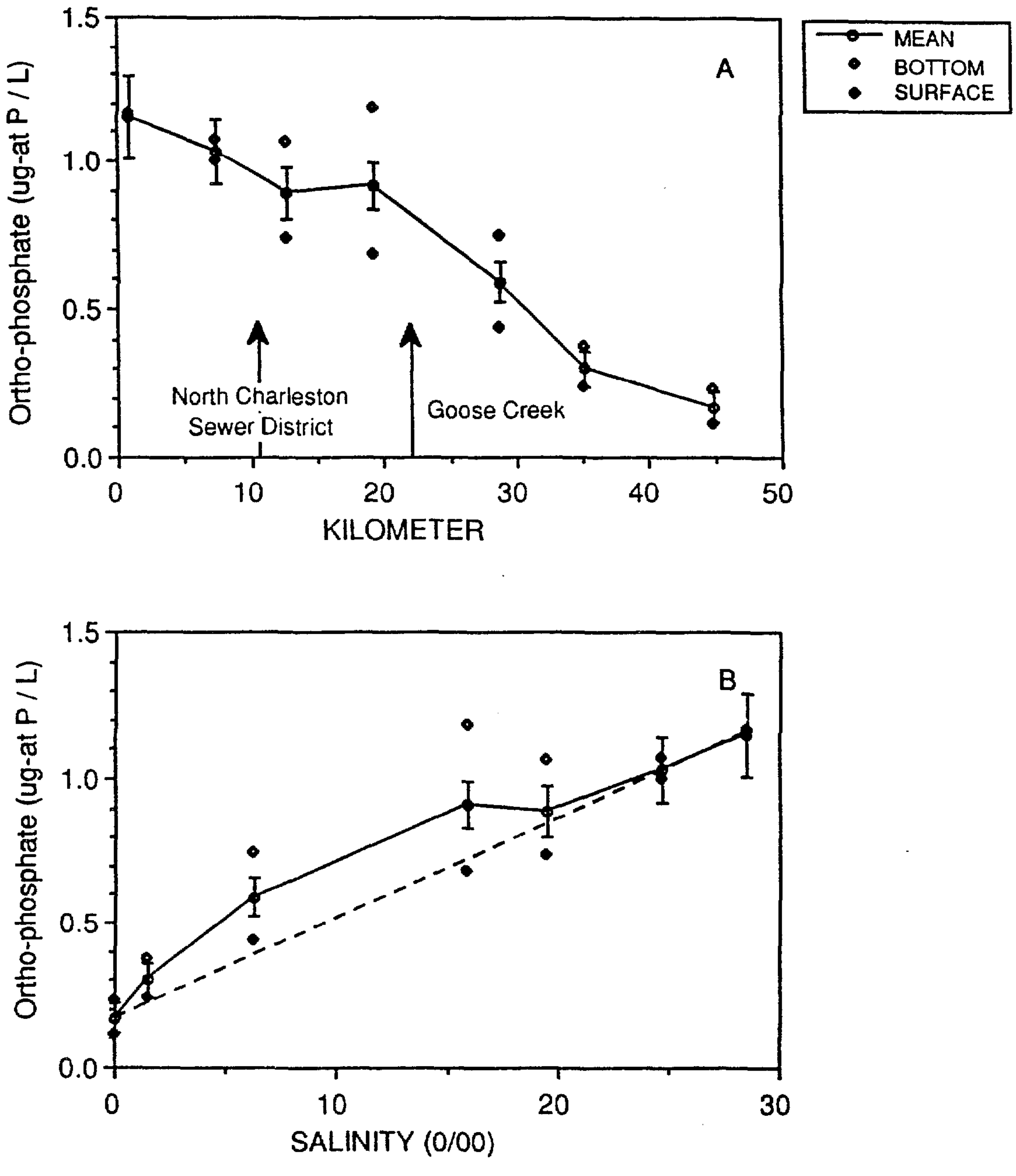


Figure IV.23. Mean ortho-phosphate concentration (ug-at P/L) for the entire sampling period Means are averages of depth, tide and month. (A) Concentration versus river km from the harbor. (B) Concentration versus mean salinity for each station along the estuarine gradient. Dashed line represents concentrations expected from conservative mixing.

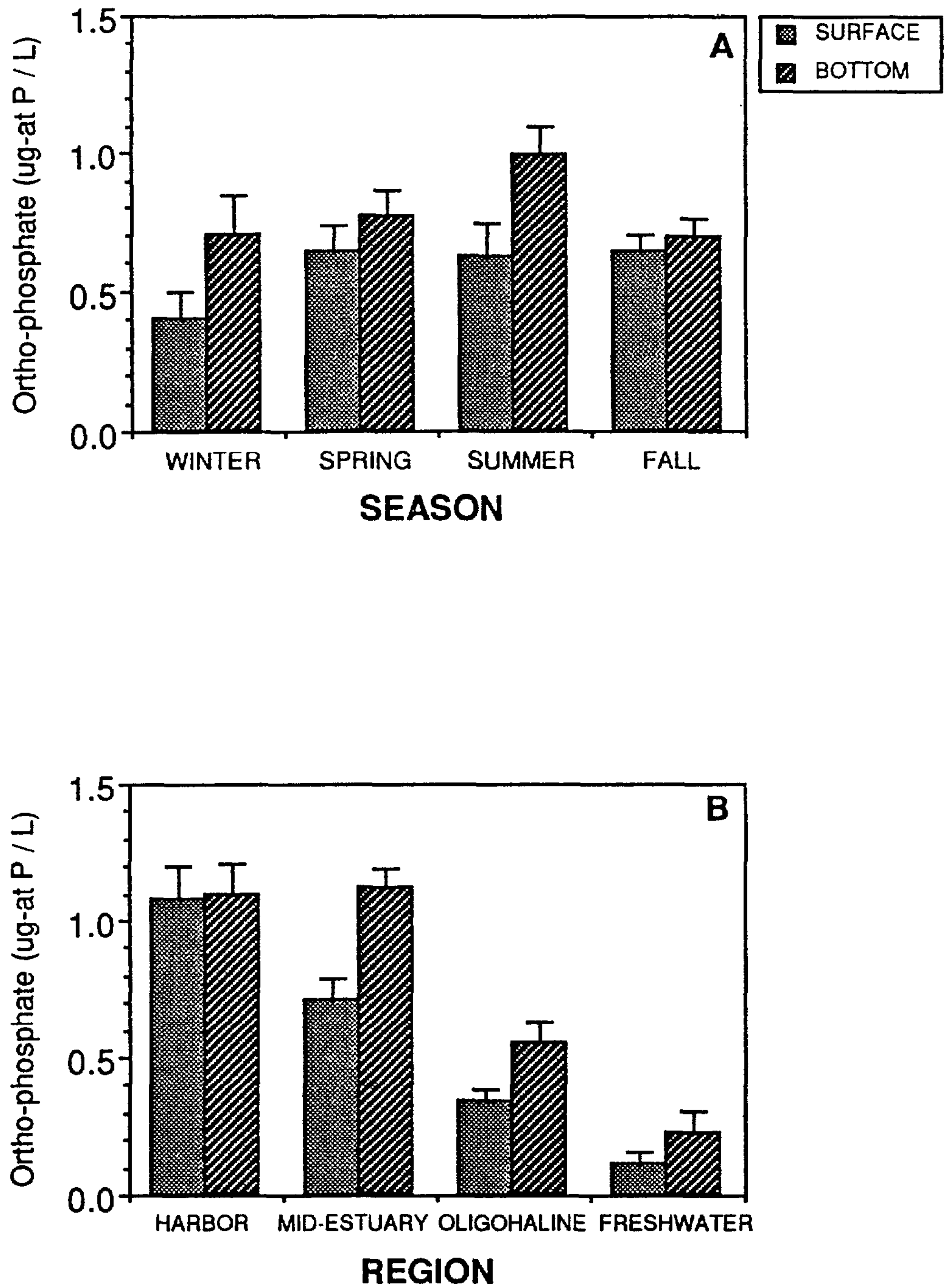


Figure IV.24. Mean ortho-phosphate concentration (ug at P/L) for surface and bottom for each season (A) or each region (B). Means are averages of tide and either station and month within a season (A) or season and station within a region (B). Error bars are standard errors.

Table IV.14. Ortho-phosphate concentrations (ug-at N/L) averaged over the seven main tributary stations for the entire sampling period by tide, depth and month.

		EXTREMES	MEAN	STANDARD ERROR	NUMBER
TIDE:	High	0-5.2	0.8	0.1	150
	Low	0-2.7	0.6	0.1	166
DEPTH:	Surface	0-5.2	0.6	0.1	158
	Bottom	0-5.2	0.8	0.1	158
MONTH:	January	0.0-2.4	0.4	0.1	28
	February	0.0-1.0	0.2	0.1	12
	March	0.0-5.2	0.9	0.2	26
	April	0.0-1.8	0.7	0.1	28
	May	0.0-1.8	0.8	0.1	28
	*June	0.0-1.6	0.7	0.1	28
	July	0.0-2.7	1.0	0.2	28
	*August	0.0-2.0	0.7	1.1	28
	September	0.0-2.1	0.8	0.2	28
	October	0.0-1.4	0.7	0.1	26
	November	0.0-1.3	0.7	0.1	28
	December	0.0-1.2	0.6	0.1	28

* Spring tide only

The seasonal changes in PO_4 concentrations varied with location in the estuary and suggested potentially different sources for the observed concentrations (Figure IV.25). In the freshwater region, concentrations were lowest during the spring and fall, with winter and summer concentrations three times higher. In contrast, the harbor spring and summer concentrations were similar (1.23 ug-at/l) and higher than winter and fall (0.93 ug-at/l). The mid-estuary seasonal patterns were similar to the harbor but at concentrations that were approximately 70% of the harbor concentrations for winter, spring and summer. No distinct seasonal pattern was evident in the oligohaline region due to apparent contributions from both the mid-estuary and freshwater regions.

The salinity mixing diagram indicated that the source for PO_4 in the Cooper River-Charleston Harbor estuarine system occurred in the lower harbor (Figure IV.23, $y = 0.277 \pm 0.032 * SAL$, $R^2 = 0.95$) and, in general, that concentrations in the Cooper River were conservatively mixed with PO_4 being transported upstream from the harbor. The correlation between salinity and PO_4 concentrations ranged from $R = 0.41$ during the winter to $R = 0.81$ during the spring. Analysis of low tide data (Table IV.15) also substantiated this conclusion. None of the riverine sources (Wando River, Goose Creek, Grove Creek or the

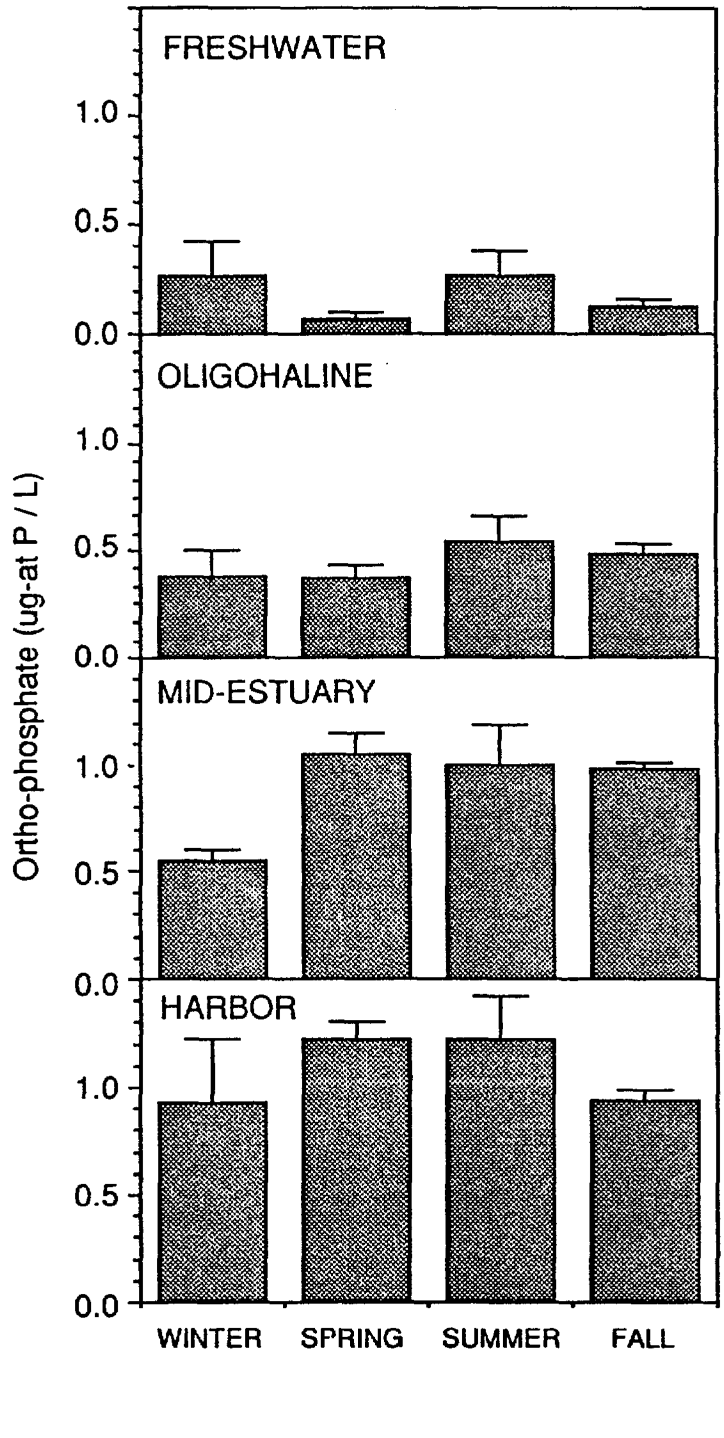


Figure IV.25. Mean seasonal ortho-phosphate concentration (ug-at P/L) and standard error by estuarine region. Means are averages of depth, tide and stations within a given estuarine region.

Table IV.15. Ortho-phosphate differences (ug-at/L) between tributary inputs and the main channel. (Low tide means averaged through the water column over the entire study period, n = 22-24, $\bar{X}t$ = tributary mean, $\bar{X}c$ = main channel mean, * indicates significant differences, p<.05)

TRIBUTARY	$\bar{X}t$	±SE	MAIN CHANNEL STA	$\bar{X}c$	±SE	Difference ($\bar{X}t$-$\bar{X}c$)
East Branch	0.1	0.04	TW	0.1	0.03	0.0
Grove Creek	0.3	0.02	J	0.3	0.06	0.0
			L	0.1	0.03	0.2*
Goose Creek	1.0	0.10	G	0.7	0.11	0.3
			H	0.6	0.10	0.4*
Wando River	1.1	0.10	C	1.1	0.13	0.0
			E	0.8	0.13	0.3*
Ashley River	1.9	0.19	A	1.3	0.12	0.6*
			C	1.1	0.13	0.8*

east and west branch of the Cooper River) contributed to concentrations observed along the salinity gradient in the Cooper River and harbor basin. No riverine source had higher concentrations than the salinity gradient station above or below the that riverine source entry point to the gradient. A minor deviation from conservative mixing occurred at approximately 6 ppt salinity with PO_4 concentrations 58% above the predicted conservatively mixed concentration but concentrations rapidly declined (within 4 ppt) to predicted values. The source of the additional PO_4 was not clear. Discharges from the NCSD may have been contributing factors, resulting in upstream transport of PO_4. Ammonium and nitrate-nitrite were positively correlated (R=0.37 and R= 0.24, respectively) with PO_4 indicating potentially similar sources.

A perplexing issue was the large deviation for PO_4 from the conservative mixing diagram substantially upstream from the ammonium or nitrate-nitrite maxima. Loder and Glibert (1980) found PO_4 increased during summer months in the Great Bay Estuary and that the increase could not be attributed to sewage or riverine inputs alone. They attributed the differences to benthic regeneration. Within the Cooper River estuary this area was also an area of high turbidity suggesting flocculation of organic material which could have contributed to benthic regeneration during summer months.

Point sources within the harbor did not appear to contribute significantly to the observed pattern (ie. Station B is not significantly different from Station A or C). The major source for PO_4 concentrations appeared to be the Ashley River. The PO_4 concentrations in the Ashley River were significantly higher (1.91 $\pm$ 0.19, 0.38-4.03 ug-at/l) than Station A (1.20 $\pm$ 0.12, 0.42-2.60 ug-at/l) or Station C (1.11 $\pm$ 0.13, ND-2.7 ug-at/l) within the harbor.

SUMMARY

1. The distributions of nutrients, organic carbon, and general water quality in the Cooper River estuary were studied along a 45 km transect from the mouth of Charleston Harbor, through industrialized urban areas, to relatively undeveloped tidal freshwater reaches. During the study period (Feb. 1988-Feb. 1989) freshwater inflow was highly variable on a daily basis (0-330 m^3/s) although seasonal fluctuations were quite moderate (117 m^3/sec, mean annual flow).

2. With reduced flows in the Cooper River (since rediversion) the distribution of salt in the estuary is less variable and less predictable in terms of freshwater input. However, the salt distribution is still significantly correlated with weekly mean flows. Up to 55% of the variability in salinity in the upper and middle reaches of the estuary can be explained by variability in river flow.

3. Surface water turbidity typically displayed two spatial peaks in the estuary. A peak in the upper reaches (30-35 km upstream) suggested increased flocculation of particulate matter at the upper zone of fresh/salt water mixing. A second peak in the harbor suggested considerable influence of turbulence and resuspension in the harbor as well as influx of highly turbid water from the Ashley River. Surface turbidity was significantly correlated with concentrations of particulate organic carbon (POC) and phytoplankton biomass.

4. Total organic matter in the estuarine waters was dominated by dissolved organic carbon (DOC) which varied largely between 1 and 10 mg/l with a mean of 4.7 mg/l. There were significant spatial trends in DOC distributions with higher concentrations in the surface water and in mid-estuarine reaches. Mixing diagrams indicated a net source of DOC within the mid-estuarine area especially in the bottom waters, perhaps related to effluent from the Westvaco paper mill.

5. Particulate organic carbon (POC) generally constituted approximately 25% of the total organic carbon and varied between 0.1 and 4.7 mg/l, with an overall mean of 1.3 mg/l. POC was composed largely of detrital material except during phytoplankton peaks when algal carbon accounted for >50% of the total POC.

Seasonal variability in POC was dominated by peaks during the winter and summer, corresponding to peaks in phytoplankton biomass. Phytoplankton biomass was typically higher in freshwater reaches and declined significantly through the estuary, showing some recovery in the harbor area. This pattern suggests a net loss of freshwater phytoplankton through the estuarine reaches and a partial succession to marine and estuarine species in the harbor.

6. Spatial patterns in total POC distribution were dominated by higher concentrations in the upper reaches of the estuary and in the bottom waters. Mixing diagrams suggested a net sink of POC from the surface waters, contributing to a bottom water source of POC in the harbor region. The sinking and decomposition of POC from the surface waters also accounts for some of the observed DOC source in the bottom waters.

7. Dissolved ortho-phosphate (PO_4) varied between <0.01 and 6.0 ug at./l with a mean of 0.7 ug at./l. There were significant spatial trends in PO4 distribution with higher concentrations in the harbor at high tide, suggesting a potential oceanic source. Concentrations varied linearly with salinity suggesting a net balance in sources and sinks of PO_4 through the estuary. Higher concentrations occurred in the bottom waters with peak concentrations occurring in March and July.

8. Dissolved inorganic nitrogen was composed of slightly higher ammonium (NH_4) concentrations than nitrate/nitrite (NO_3-NO_2) concentrations. Ammonium ranged from <0.01 ug at./l to 126.5 ug at./l with a mean of 3.78 ug at./l. Higher concentrations were detected during low tide and in surface waters at mid-estuary stations. Mixing diagrams suggest a major source in the mid-estuarine reaches, apparently dominated by ammonium-rich discharges from the North Charleston waste water treatment facility. Higher concentrations occurred during April and July.

9. Similar patterns were observed for nitrate-nitrite with highest concentrations at mid-estuary stations in surface waters. Mixing diagrams suggest a source in the mid-estuarine reaches. Peak nitrate-nitrite concentrations were typically located one station below peak ammonium concentrations at low tide and one station above at high tide suggesting rapid oxidation of the ammonium entering the estuary in the mid-estuarine reaches. Unlike ammonium, there were no significant differences between high and low tide. Highest nitrate-nitrite concentrations occurred during the late summer (August and September).

CHAPTER V

PHYSICAL DYNAMICS

by

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INTRODUCTION

Background and Objectives:

Within the past four decades, the Charleston Harbor estuary has experienced more far-reaching impacts from man-made changes than have occurred in most other estuarine systems. The 1942 river flow diversion increased the discharge of the Cooper River fifty-fold. This action, coupled with a deepening of the harbor navigation channel two years later, caused excessive shoaling and sedimentation within the harbor. To counteract this problem, rediversion of approximately 70% of the Cooper River discharge away from the estuary occurred in 1985.

Major freshwater flow alterations have direct bearing on both salinity and tidal current regimes in an estuary. For example, Ingram *et al.* (1986) found that a 50% increase in freshwater discharge into the Eastmain River estuary caused a significant drop in both salinity and tidal currents in the estuary, as well as an increase in suspended sediment load and erosion. When the freshwater flow was subsequently reduced by 80%, a gradual salinity intrusion occurred in response to strong tidal and wind forcing rather than freshwater flow changes (Lepage and Ingram, 1986).

Circulation and salinity regimes became more variable as well. For example, Sharp *et al.* (1986) found that salinity stratification in the Delaware Estuary during low discharge was also predominantly driven by winds rather than freshwater flow. High freshwater flow into an estuary acts as the dominant control on salinity and tidal currents, but under low flow conditions, other factors, such as winds and more pronounced tidal currents become the dominant controls.

Kjerfve and Magill (1990) analyzed a 45-year record of salinity and discharge from Charleston Harbor and found a strong relationship between the two during the high freshwater input period subsequent to river diversion. But since the freshwater flow has been reduced due to rediversion, it has been unrelated to salinity variations because variability in monthly freshwater discharge is now only minor. Instead, far-field forcing from the adjacent coastal ocean has been shown to cause upstream propagation of estuarine waters (Rutz, 1987) and an upstream relocation of the 1 ppt isohaline by 15-20 km.

There are also ecological consequences of altering freshwater flow into an estuary. Both Sharp *et al.* (1986) and Bennett *et al.* (1986) found that high freshwater discharge stimulates primary productivity and increases particulate nutrient loading. Decreased freshwater flow and the resultant upstream salinity intrusion can alter the location of nursery habitats of larval shellfish, which are often of great commercial importance, as in the case of the *Penaeus* shrimp fishery in Charleston Harbor. Freshwater flow alterations also affect the species distribution of marine and brackish water marsh vegetation (Bradley *et al.*, 1990).

The objective of this study was to develop the capability to diagnose estuarine responses to changing freshwater flow. This was accomplished by (1) implementing a numerical model of the Charleston Harbor estuary and tidal portions of the Cooper, Ashley and Wando Rivers; (2) simulating estuarine currents, water level variations, and salinity changes; and (3) synthesizing existing salinity and tidal elevation data and new oceanographic data collected during two intensive post-rediversion sampling programs.

Review of the Diversion and Rediversion Projects:

Prior to 1942, the combined discharge of the Ashley, Cooper, and Wando Rivers was only 10 m^3/s (US Army Corps of Engineers, 1966), circulation in the harbor was tidally dominated, and the salinity structure was well-mixed. In the late 1930's, an increasing demand arose for hydroelectric power, and the Santee River, with a freshwater discharge of 525 m^3/s, was chosen as a suitable power source. The South Carolina Public Service Authority (SCPSA) initiated the Santee-Cooper Hydroelectric Power Project, with the goal of diverting 88% of the Santee river flow into the Cooper River for power generation at Pinopolis.

The diversion project was completed in 1942, resulting in the construction of (Figure V.1): 1) Wilson Dam on the Santee River, creating Lake Marion; 2) Pinopolis Dam on the Cooper River, creating Lake Moultrie; and 3) a 12 km long diversion canal between Lake

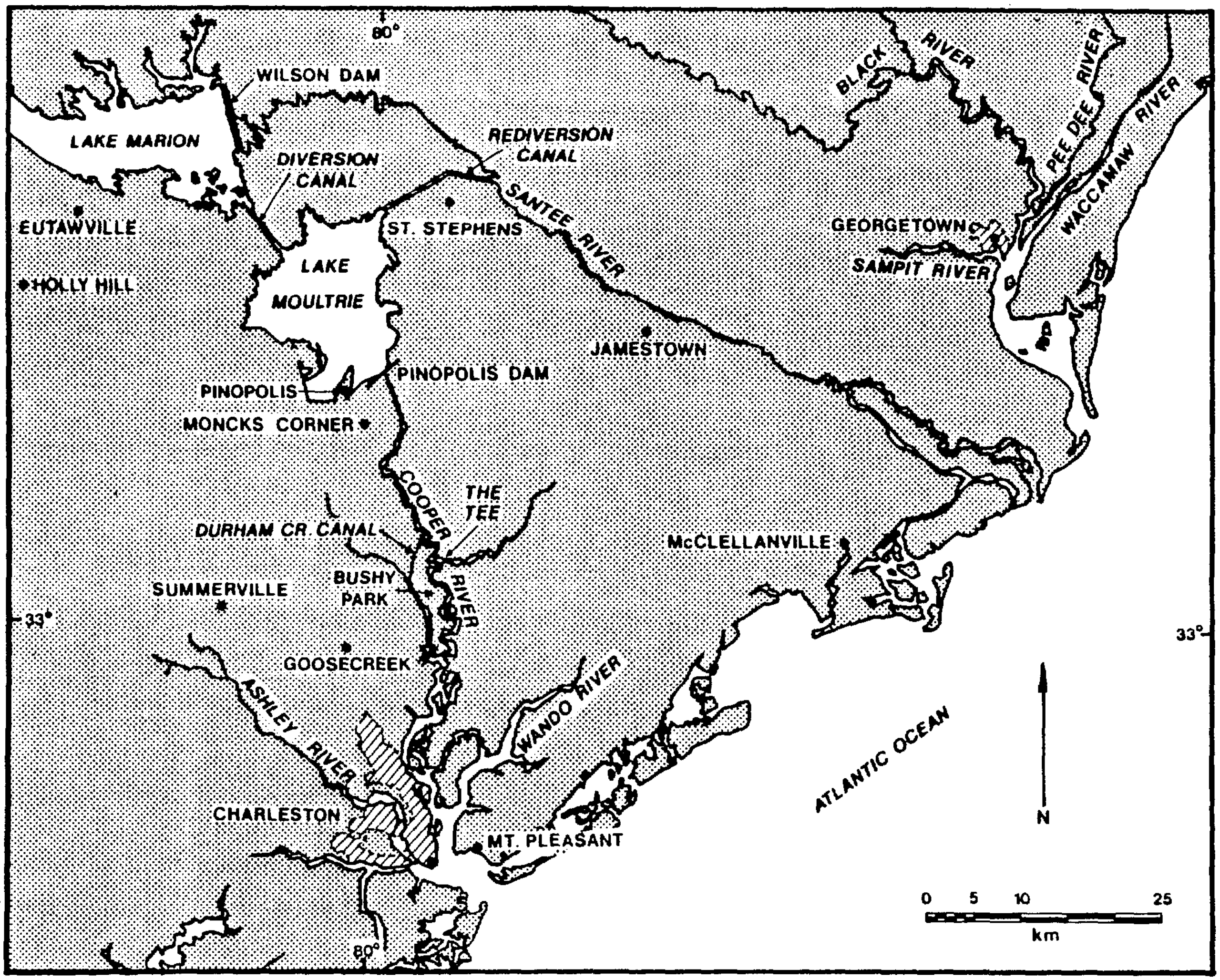

Figure V.1. Area map of the Charleston Harbor/Cooper River system.

Marion and Lake Moultrie, through which the Santee River flow was diverted into the Cooper River (Kjerfve, 1976). The mean Santee River discharge subsequently decreased to 62 m^3/s (SC Water Resources Commission, 1979), while the mean Cooper River discharge increased to 442 m^3/s (Kjerfve, 1976).

As a consequence, the mean salinity in Charleston Harbor decreased from 30.0 ppt to 16.8 ppt (Zetler, 1953), gravitational circulation became the dominant estuarine circulation mode (Kjerfve, 1976), and the estuarine salinity structure became partially mixed. These changes lead to increased shoaling in the harbor and the need for frequent maintenance dredging of the ship channel. Prior to diversion, the harbor ship channel had been self-maintaining, and the estuary as a whole was slowly deepening (Simmons and Herrmann, 1972).

Two years after diversion, the ship channel was deepened from 9.1 m to 10.6 m. This modification lead to a further strengthening of gravitational circulation, which compounded the shoaling problem by causing greater landward transport of marine sands. By the late 1970's, shoaling and sedimentation had become so excessive that the Army Corps of Engineers was spending over $5 million a year in maintenance dredging (Little, 1974b).

To alleviate shoaling in the harbor but still maintain hydroelectric power potential, the SCPSA carried out the Santee-Cooper Rediversion Project. A major portion of the Cooper River flow was to be rediverted back to the Santee in the hopes of decreasing the gravitational circulation and thus the landward migration of bed materials. The rediversion project was completed in 1985, with the main construction feature being an 18.5 km long canal from Lake Moultrie to the Santee River (Figure V.2). Most of the upper Santee River flow is now being channeled into Lake Moultrie, through the rediversion canal, and back to the lower Santee via the St. Stephen's rediversion canal (Figure V.1).

The SCPSA now maintains the Cooper River flow at the Pinopolis Dam to a monthly average of 122 m^3/s. This rate is high enough to sustain hydropower generation, and at the same time ensure that no marine waters enter the Durham Creek Canal and thus contaminate the Bushy Park reservoir. Whether or not this flow is low enough to sufficiently reduce gravitational circulation in the harbor remains to be validated.

The United States Geological Survey (USGS) maintains eight water level/water quality gauges along the Cooper River and Durham Creek Canal, from which they constantly monitor conductivity and advise the SCPSA of any need to increase the discharge at Pinopolis Dam to prevent salinity intrusion into Durham Creek Canal. This is to satisfy several industries located in Bushy Park who use freshwater from the Back River Reservoir (Figures V.1, V.2) for industrial operations, as well as the city of Charleston, which uses the reservoir as a municipal water supply.

METHODS

Intensive field sampling was conducted in Charleston Harbor during two periods: 20 April - 17 August 1987, and 8 February - 25 March 1988. The measurements obtained include (1) vertical profiles of conductivity, temperature, density (CTD), and transmissivity collected along a longitudinal transect from Fort Sumter to Pimlico on the Cooper River; (2) vertical profiles of current velocity, conductivity, temperature, density, and transmissivity collected along a cross-sectional transect of the harbor mouth from Fort Sumter to Fort

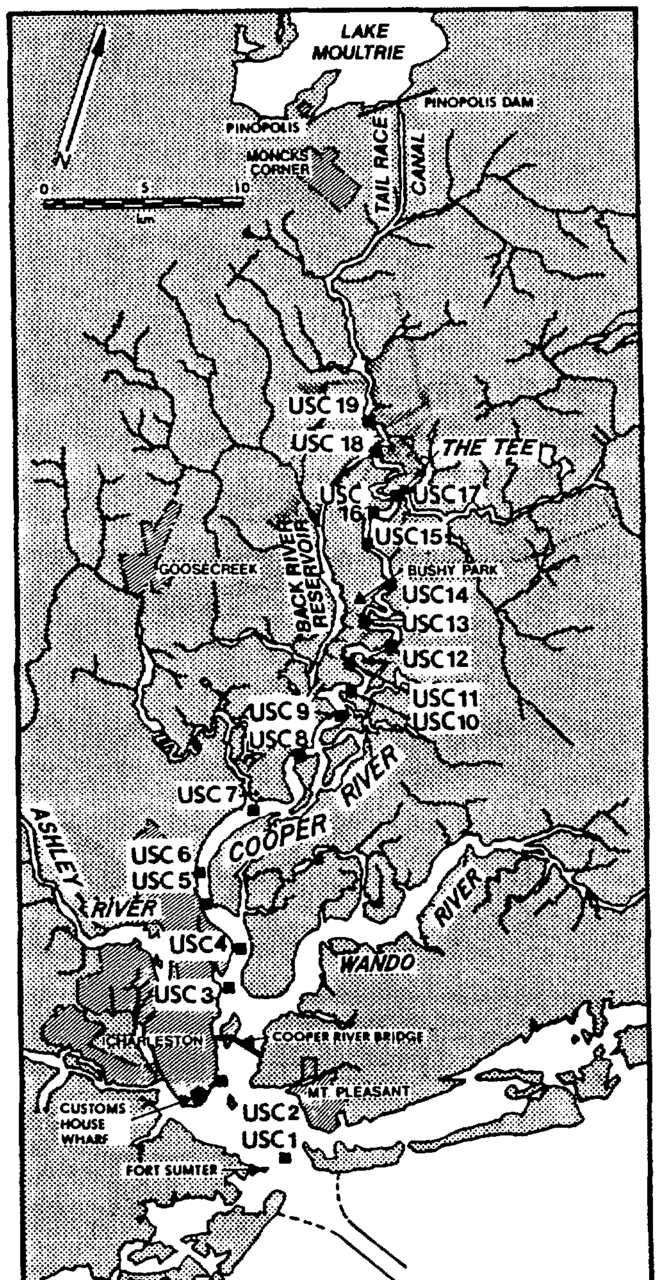

Figure V.2. Location of the CTD stations in Charleston Harbor and the Cooper River.

Moultrie; and (3) time series of current velocity, conductivity, temperature, and water elevation from an InterOcean S4 electromagnetic current meter mooring located upstream on the Cooper River. The vertical CTD and transmissivity profiles were made using a SeaBird CTD/Sea Tech transmissometer instrument, and vertical current velocity profiles along the cross-section were made using current vanes (Kjerfve and Medeiros, 1989).

Both sampling programs were coordinated with field work carried out simultaneously by the National Ocean Service (NOS)/NOAA. NOS scientists collected vertical profiles of current velocity using two self-recording bottom-mounted RD Instruments Acoustic Doppler Current Profilers (ADCPs). One ADCP was moored near Fort Sumter, and the other unit was moored at alternate locations within the harbor.

The USGS Water Resources Division office in Columbia, SC, maintains eight permanent water quality stations along the banks of the Cooper River. Water level, conductivity, temperature, dissolved oxygen, and pH are monitored regularly at the stations every 10, 15, or 60 minutes, depending on the station and parameter being measured. These data are telemetered in real-time from the station to the geostationary GOES satellite, and are from there beamed to the USGS/WRD office in Columbia, SC. Through a mutual agreement between the South Carolina Sea Grant Consortium, USGS, and the University of South Carolina, we had real-time access to these data via computer modem. We downloaded salinity data from the six stations for the duration of the first sampling period, 20 April -17 August 1987.

In addition, time series of water elevations were obtained from the NOAA station at the Customs House Wharf for the time periods 20 April - 14 June 1987, and 1 February - 2 April 1988. These data were used for tidal analysis during the two sampling periods.

Longitudinal Salinity Measurements:

To characterize longitudinal-vertical salinity distributions in the Charleston Harbor estuary, we selected 19 stations from the harbor mouth to Pimlico, which is located 62.6 km upstream on the Cooper River from Fort Sumter (Figure V. 2). Vertical profiles of conductivity, temperature, density, and light transmissivity were measured at each station during the two sampling periods: JD 117 - JD 177 in 1987, and JD 39 - JD 85 in 1988. The sampling followed the upstream propagation of high tide, beginning at station 1 near Fort Sumter and ending approximately 2.5 hr later at station 19 near Pimlico. For each sampling period, 15 days of high quality data were obtained. The SeaBird CTD unit recorded data at a frequency of 24 scans per second. Three consecutive scans were averaged to reduce the sampling frequency to 8 samples per second.

Cross-Sectional Measurements:

Vertical CTD, transmissivity, and current velocity profiles were measured from three stations across the harbor mouth from Fort Sumter to Fort Moultrie. The simultaneous collection of CTD and current velocity data was necessary for estimating water and salt fluxes between Charleston Harbor and the adjacent coastal ocean.

Cross-sectional sampling was carried out over 2 neap tidal cycles on 5-6 May 1987, 2 spring tidal cycles on 15-16 June 1987, and 2 neap tidal cycles on 18-19 February 1988. Some of the velocity data were contaminated (especially on 19 February 1988) because of strong tidal currents and the boat dragging the anchor.

Velocity, Tide, and Flow Measurements:

One current meter instrument mooring was successfully deployed in the Cooper River on 20 April 1987. The mooring consisted of a Sensordata SD2000 current meter with temperature sensor near the water surface and an InterOcean S4 electromagnetic current meter with conductivity, temperature, and pressure sensors near the bottom. The mooring was deployed 26.1 km upstream on the Cooper River from Fort Sumter. It was recovered on 29 June 1987, yielding high quality time series data.

Because of the importance of current velocity time series in assessing current flows, a third current meter instrument mooring, consisting of two S4 current meters, was deployed at river km 26.1 for the period 16 July - 17 August 1987. The instruments were positioned 3.5 m apart in 11.5 m of water. They measured current speed and direction, temperature, conductivity, and water elevation. The instruments were programmed to average 0.5 Hz measurements into 2-min averages every 10 minutes. The time series data obtained from both instruments are of excellent quality and have been combined with the USGS data to assess the dynamics and salinity variability of the Cooper River.

Daily estimates of Cooper River discharge were obtained from the South Carolina Public Service Authority, Santee-Cooper Power Utility, who is in charge of regulating the flow release from the Pinopolis Dam. The daily averaged discharge was 140 m^3/s (±69) during the 1987 field sampling and 126 m^3/s (±66) during the 1988 sampling (Figure V. 3).

Data Reduction:

A total of 209 CTD profiles were obtained from the 19 stations along the Cooper River during the 1987 sampling, and 163 during the 1988 sampling. Each profile was

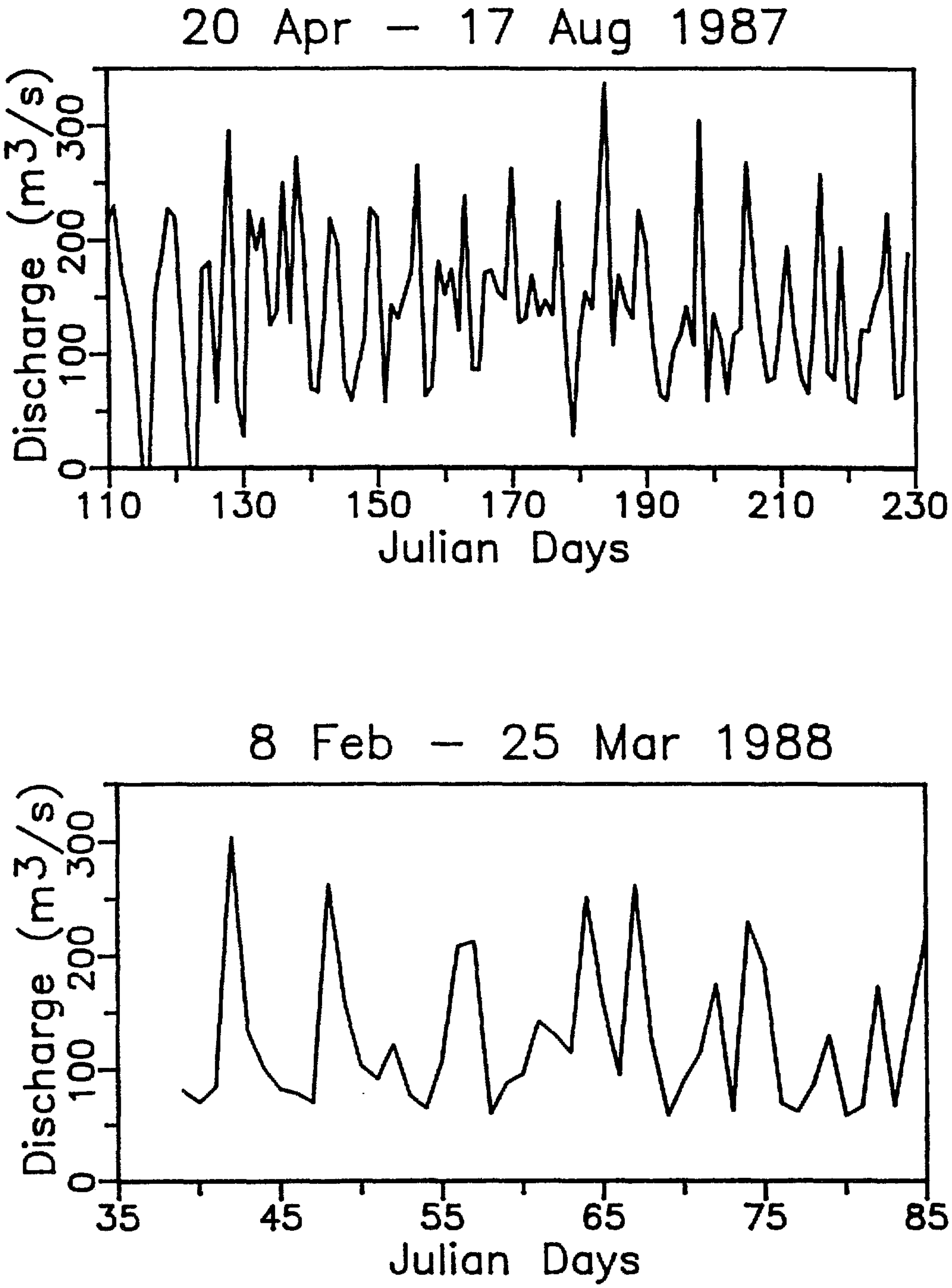

Figure V.3. Freshwater release over the Pinopolis Dam during the 1987 and 1988 field sampling periods.

converted to ASCII-coded engineering units using SeaBird Seasoft software, then edited. Data contaminated by mud-fouling, surface readings recorded before the instrument was submerged, and transmissivity readings altered by reflected sunlight were deleted from the records.

The data points in each edited profile were averaged over 10 cm distances (bins). The 10-cm averaged data were used to construct vertical profiles of salinity, density, transmissivity, and temperature at each station. These data were further averaged into 0.5 m depth increments, then interpolated into 1 m (vertical) by 1 km (horizontal) grids for the sake of contructing the graphics. The mean and root-mean-square (rms) salinity and transmissivity distributions were also calculated and plotted.

The CTD, transmissometer, and current velocity data from the cross-sectional sampling were edited, averaged into 10 cm increments, and averaged again into 0.5 m depth increments. Depths at each station were adjusted for actual tidal elevation using values interpolated from the hourly data collected at the NOAA/USGS tide gauges at the Customs House Wharf. The adjusted depths were used to construct time-weighted mean depths for each station. Cross-sectional plots were then produced for salinity, transmissivity, temperature, density, velocity, and salt flux. The cross sections were oriented looking downstream with Fort Moultrie to the left.

Time series data from the two S4 current meters moored in the Cooper River during the 1987 sampling were downloaded to a microcomputer, referenced to Greenwich Mean Time (GMT), and converted into uniform ASCII-coded metric units. Statistical analyses were then performed on the time series of current velocity, conductivity, temperature, and water elevation data obtained from the instruments.

Harmonic analysis was performed on the current velocity records from the two moored S4 current meters and the water elevation record from the near-bottom S4 meter. Fifteen-day rather than 29-day records were chosen to eliminate any effects of biofouling.

In addition, an excellent tidal record was obtained over the 4-month 1987 sampling duration from the USGS tide gauge at the Customs House Wharf. Harmonic analysis was performed on this record as well. We also analyzed time series of conductivity and temperature obtained over the 1987 sampling program from three USGS stations: General Dynamics (near USC station 13), Dean Hall (near USC 16), and Pimlico (near USC 19) (Figure V.2). These were the only stations having reasonably complete time series records. The conductivity and temperature data were converted to salinity (psu) and presented as time series plots (Figure V.4).

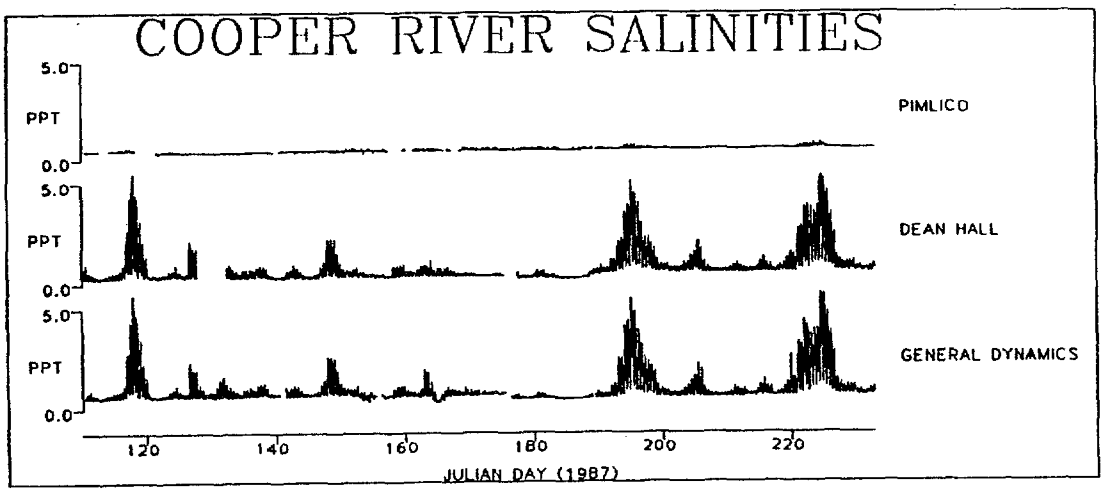

Figure V.4. Time series of salinity computed from conductivity and temperature data recorded at the USGS stations at Pimlico (62.6 km from Fort Sumter), Dean Hall (51.1 km), and General Dynamics (41.9 km).

The hourly USGS time series of water elevations obtained from the Customs House Wharf during the 1988 field sampling were low-pass filtered using an 8th-order, recursive Butterworth filter with a 36-hour cut-off frequency. Statistical analysis was performed on the hourly low-passed filter data.

Numerical Modeling: Circulation and Dispersion:

A coupled two-dimensional circulation/dispersion model for the Charleston Harbor estuary, including the Ashley and Wando rivers was implemented to assess and simulate tidal currents, water levels, and salt distributions. A separate one-dimensional circulation/dispersion model of the estuarine portion of the Cooper River was linked to the two-dimensional model.

The two-dimensional model consists of 2,312 active 300 x 300-m grid elements. It is vertically-integrated and is solved explicitly by finite-difference equations using a leap-frog scheme in time and a staggered central difference parameter representation in space. The finite difference equations are second order accurate in space and first order accurate in time. The model is time-varying, utilizes a Manning's n parameter to simulate bottom friction, and assumes horizontal dispersion to be proportional to the local instantaneous velocity and the grid spacing. The model is forced by prescribed tidal constituent amplitudes and phases at the open ocean boundary, freshwater discharge at one or more locations within the model, and temporally and spatially varying wind stress.

The formulation of the circulation model (see Kjerfve *et al.*, 1988) is a modification from Blumberg (1977a), with the model starting point being the set of global shallow water equations (Welander 1957) resulting from vertical integration of the momentum balance equations. The equations are time-varying, retain field acceleration terms, include Coriolis effects, approximate pressure terms by barotropic pressure gradients, incorporate quadratic bottom friction, and include wind stresses. Blumberg (1977b) and Tee (1976) found that any simplification of the governing equations produced major changes in simulated circulation patterns. The model uses a right-handed Cartesian coordinate system. The vertically integrated x-and y-component equations are:

$$\partial uH/\partial t + \partial u^2 H/\partial x + \partial uvH/\partial y - fvH + gH\partial\eta/\partial x = \tau_x - ku(u^2 + v^2)^{1/2} \qquad (1)$$

$$\partial vH/\partial t + \partial uvH/\partial x + \partial v^2 H/\partial y + fuH + gH\partial\eta/\partial y = \tau_y - kv(u^2 + v^2)^{1/2} \qquad (2)$$

where u and v are vertical averages of depth-varying velocities u' and v', respectively, i.e.,

$$u = \int u'dz/H \qquad (3)$$

$$v = \int v'dz/H \qquad (4)$$

where the limits of integration are from the bottom, $z = -h$, to the water surface, $z = \eta$; H is the local water depth, $h + \eta$; k is a friction coefficient, expressed using Manning's n; f is the Coriolis parameter, expressed as

$$f = 2\Omega\sin\Phi \qquad (5)$$

where Ω is the rotation of the earth (7.29×10^{-5} s^{-1}) and Φ is latitude. The model incorporates the β-plane approximation to account for variations in the Coriolis parameter with latitude,

$$f = f_o + \beta(\Phi + \Phi_o)R \qquad (6)$$

where f_o is the Coriolis parameter at the lowest latitude in the grid; Φ is the lowest latitude in the grid; and R is the radius of the earth, 6.37×10^6 m.

Total wind stress, τ, is computed from the 10 m wind speed, W_{10},

$$\tau = \rho_a C_{10}(W_{10})^2 \rho^{-1} \qquad (7)$$

and is decomposed into x- and y-component 10 m wind stresses,

$$\tau_x = \rho_a C_{10} W_u | W_u | \rho^{-1} \tag{8}$$

$$\tau_y = \rho_a C_{10} W_v | W_v | \rho^{-1} \tag{9}$$

where ρ is water density, 1020 kg m^{-3}; ρ^a is air density, 1.2 kg m^{-3}; and C_{10} is a non-dimensional drag coefficient, 1.6 x 10^{-3}, assumed independent of atmospheric stability and wind speed.

In addition to momentum, the model conserves mass. Because the model is designed for shallow water systems, it is valid to assume density is not affected by pressure. The general expression for conservation of mass is then replaced by the continuity equation. Vertical integration of the continuity equation from the bottom to the water surface yields

$$\partial\eta/\partial t + \partial uH/\partial x + \partial vH/\partial y = 0 \tag{10}$$

Using only barotropic components to represent pressure gradients in the momentum component equations assumes that water density is vertically homogeneous and that longitudinal pressure gradients due to density variations are negligible (Blumberg, 1978). This is a simplification, but in a tidal situation, barotropic terms dominate the baroclinic terms.

Further, vertical integration causes information on the vertical velocity structure to be lost, precluding calculation of longitudinal-vertical estuarine circulation. In most systems, however, baroclinic pressure gradient terms are negligible, and density-driven estuarine circulation is highly variable and of secondary importance in comparison to residual tidal circulation.

The circulation model incorporates a Manning's n in the friction formulation. In a steady, uniform velocity field, specific friction force per unit wetted area, K, is balanced by gravity, g, acting in the downslope direction (Le Méhauté, 1976). This balance is expressed as

$$K = gR_h S_f \tag{11}$$

where R_h, the hydraulic radius, is the cross sectional area, A_{cs}, divided by the wetted perimeter, Γ,

$$R_h = A_{cs}/\Gamma \approx H \tag{12}$$

and S_f is the surface friction slope. S_f is defined using Manning's n,

$$S_f = n^2 u^2 R_h^{-4/3} \tag{13}$$

(Chow. 1959). Assuming a fully turbulent flow field, K can also be expressed as

$$K = ku^2 \tag{14}$$

By equating eqs. (11) and (14), substituting eq. (13) for S_f, and using H in lieu of R_h, the friction coefficient can be expressed in terms of Manning's n and local depth by

$$k = gn^2 H^{-1/3} \tag{15}$$

The dispersion model allows calculation of concentration distributions as a function of time within the model domain. It assumes that the constituents to be modeled behave conservatively. At present, we have only simulated salinity concentrations for Charleston Harbor.

Conservation of mass applied to a conservative constituent, C, is expressed

$$\partial CH/\partial t + \partial uCH/\partial x + \partial vCH/\partial y - \partial HK_x \partial C/\partial x^2 - \partial HK_y \partial C/\partial y^2 = F \tag{16}$$

where F is the total flux of the constituent into or out of the modeled domain and includes fluxes through the air-sea interface, the sediment-water column interface, exchanges with the adjacent ocean, and input from rivers.

The dispersion equation employs effective dispersion coefficients, K_x and K_y, which are functions of velocity and grid size,

$$K_x = 0.2\ \Delta x(u^2 + v^2)^{1/2} \tag{17}$$

$$K_y = 0.2\ \Delta y(u^2 + v^2)^{1/2} \tag{18}$$

The choice of dispersion coefficients is critical and was made through iterative comparisons of model calculations to field data.

The one-dimensional coupled circulation/dispersion river model also uses a vertically integrated, finite difference formulation. The model consists of 96 active 300 x 300-m grid elements for the Cooper River. The governing equations are the same as those used in the two-dimensional model. If we define cross-sectional river discharge per unit width, q, as

$$q = uH \quad \text{or} \quad q = vH \tag{19}$$

then conservation of momentum applied to the one-dimensional, vertically integrated model is

$$\partial q/\partial t + \partial (q^2/H)\partial r + gH\partial \eta/\partial r = \tau_r - k(q/H)|q/H| \tag{20}$$

The one-dimensional vertically integrated continuity equation is written

$$\partial \eta/\partial t + \partial q/\partial r = 0 \tag{21}$$

where r is distance measured along the river. Likewise, the one-dimensional dispersion equation is

$$\partial CH/\partial t + \partial qC/\partial r - \partial HK_r\partial C/\partial r^2 = F \qquad (22)$$

The linked two-dimensional/one-dimensional model achieves quasi-steady state conditions after three tidal cycles for circulation modeling and after 20-25 cycles for dispersion modeling. The model has been calibrated with velocity, tidal elevation, and salinity data collected both from the USC field sampling and the USGS database.

Time-Varying Boundary Formulation:

Tidal flooding of the marsh wetlands is a significant factor that must be accounted for in modeling the Charleston Harbor estuarine system. In the two-dimensional model we have thus incorporated an algorithm to allow water/land boundaries to vary in time as a function of fluctuations in free surface elevations due to tidal flooding and ebbing.

The flooding routine for computation of velocity and water elevations is modified after Flather and Heaps (1975). Their model allowed marsh elements to go completely dry during ebb tide and to resubmerge during flood tide. Our model, however, stops the computations when the water level has fallen below a minimum threshold depth, which we have set to be 10 cm. Setting this threshold depth is necessary because water depth appears as a variable in the denominator of the momentum balance equations. Thus, if the water level were allowed to drop to zero, the model becomes numerically unstable, and we have determined that at depths below our set threshold depth of 10 cm, the computed velocities become unrealistically high. We have shown through computations that the 10-cm of water retained on the marsh at low tide represents an insignificant fraction of the total tidal prism. In fact, with the addition of the time-varying boundary routine, the simulated tidal prism is now similar in magnitude to the actual tidal prism.

RESULTS AND DISCUSSION

Since 1985, the monthly averaged Cooper River discharge has been maintained at a steady 122 m^3/s by regulation of the release at the Pinopolis Dam. Although daily discharge variations have been considerable (e.g., Figure V.3), the monthly variations have been small. Still, the extent of the estuarine zone varied substantially along the Cooper River, largely due to far-field forcing from the coastal ocean and the lower harbor, rather than due to the freshwater flow (Rutz, 1987; Kjerfve and Magill, 1990). This is clearly exemplified by the time series of salinity variations at three Cooper River locations (Figure V.4).

Choosing the 1 ppt isohaline as a convenient indicator, the upstream limit of the estuarine zone on the Cooper River varied from 36 to 45 km from Fort Sumter at high tide during the study periods. The longitudinal salinity data have been summarized (Table V.1) for every 5 km from Fort Sumter and presented as depth-averaged values. Tidal range (or spring vs. neap tide) does not appear to be a particularly good indicator of the upstream extent of the estuarine zone. For example, at 30 km, the highest salinity occurred during the smallest tides. Thus, contrary to expectations, the salinity did not intrude farther upstream on spring tides. As the freshwater discharge was approximately constant, we conclude that far-field forcing from the coastal ocean and lower estuary, due to meteorological forcing, is an important factor in driving the salinity farther upstream on the Cooper River.

Charleston Harbor and the Cooper River experience a semidiurnal tide with a 1.8 - 2.0 m tidal range. High water propagates upstream from the Customs House Wharf to the Durham Creek Canal entrance in approximately four hours. The tidal range does not

Table V. 1. Results of 31 depth-averaged longitudinal salinity profiles, 1987-1988, including the mean salinity (S), standard deviation (SD), and number of samples (n) for each tidal range. (-) signifies a salinity of < 1 ppt.

Tidal		Kilometers upstream from Fort Sumter										
Range	0	5	10	15	20	25	30	35	40	45	50	55
1.9-1.6 m												
S	34.4	31.0	27.3	22.8	19.2	14.1	9.4	4.7	3.4	1	1	1
SD	1.1	2.7	1.7	1.7	1.6	1.7	2.6	2.9	2.6	-	-	-
n	7	7	7	6	6	7	7	7	7	7	7	7
1.3-1.5 m												
S	33.8	30.4	26.9	22.8	19.3	15.1	9.8	3.4	1.7	1	1	1
SD	1.2	1.7	1.4	2.0	1.5	2.0	2.0	1.8	1.3	-	-	-
n	9	9	9	9	9	9	9	9	9	9	9	9
1.0-1.2 m												
S	32.8	29.0	22.9	23.2	20.7	16.9	12.1	4.7	1.9	1	1	1
SD	1.4	1.6	8.0	1.6	1.6	1.7	2.6	3.1	1.5	-	-	-
n	12	13	13	12	12	12	12	12	12	12	12	12

change appreciably over this 54-km distance. Although the tidal water level and tidal velocity are approximately 90° out of phase in the lower harbor as can be expected for a standing tidal wave, the phase relationship changes along the extent of the estuary, presumably as a consequence of frictional influence. Vertical stratification in Charleston Harbor is highly variable. During spring tides, when tidal currents may reach 2 m/s and the mixing intensity is great, the Charleston Harbor and Cooper River are well mixed vertically, as can be seen in Figure V.5a. The strong tidal currents and bottom-generated turbulence break down the density stratification and cause the estuary to be less horizontally stratified. On the other hand, substantial vertical stratification develops in the Cooper River during neap tides, particularly between river km 15 and 30, when currents and mixing intensity are less developed (Figure V.5b). The stratification parameter (Hansen and Rattray, 1966), $\Delta S/S_0$, measured 0.68 (±0.23) between river km 15 and 30 during neap tides, but only 0.23 (±0.12) during spring tides.

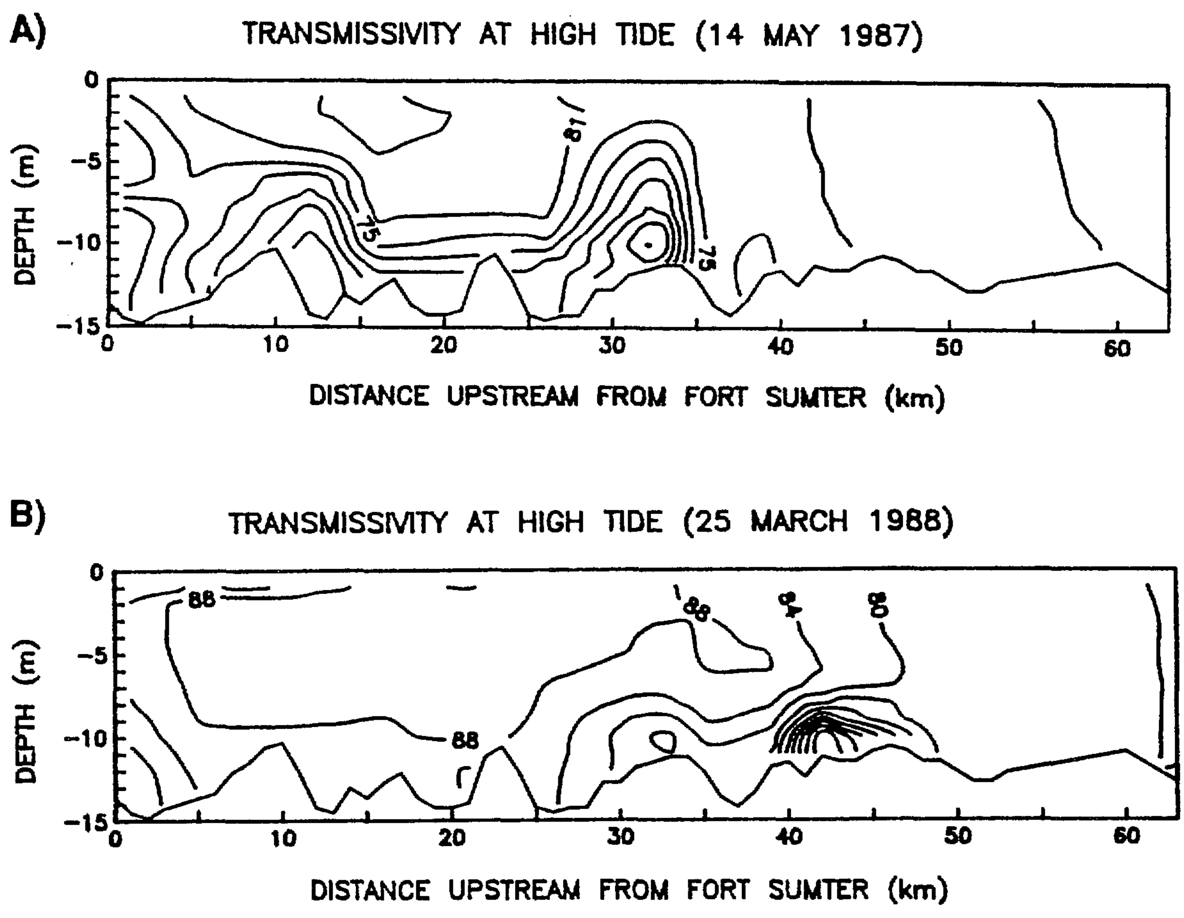

Figure V.5. High tide transmissivity distribution along the Cooper River from Fort Sumter (0 km), showing the turbidity maximum zone during (A) a spring tide and (B) a neap tide.

The transmissivity profiles collected along the estuary (Figure V.6) demonstrate the occurrence of a well defined turbidity maximum zone 20 to 35 km upstream from Fort Sumter. The zone is characterized by salinities from 7 to 17 ppt, 50% or less light transmissivity over a 5 cm path-length, and total suspended solids (TSS) concentrations from 50 to 100 ppm. Typical TSS concentrations further upstream or downstream measure only from 20 to 60 ppm. The turbidity maximum zone varies horizontally along the Cooper River due to far-field forcing in a fashion similar to the variations in the salinity distribution. In addition, the turbidity maximum zone (and also the salinity distribution) oscillates 7 to 12 km upstream-downstream over a typical neap and spring tidal cycle, respectively. Maximum upstream penetration of both the turbidity maximum zone and the salinity distribution coincides with high tide.

The cross-sectional measurements from Fort Sumter to Fort Moultrie have yielded an interesting, high quality data set, consisting of hourly time series of salinity, velocity, transmissivity, sediment concentration, density and temperature for 2 tidal cycles. The

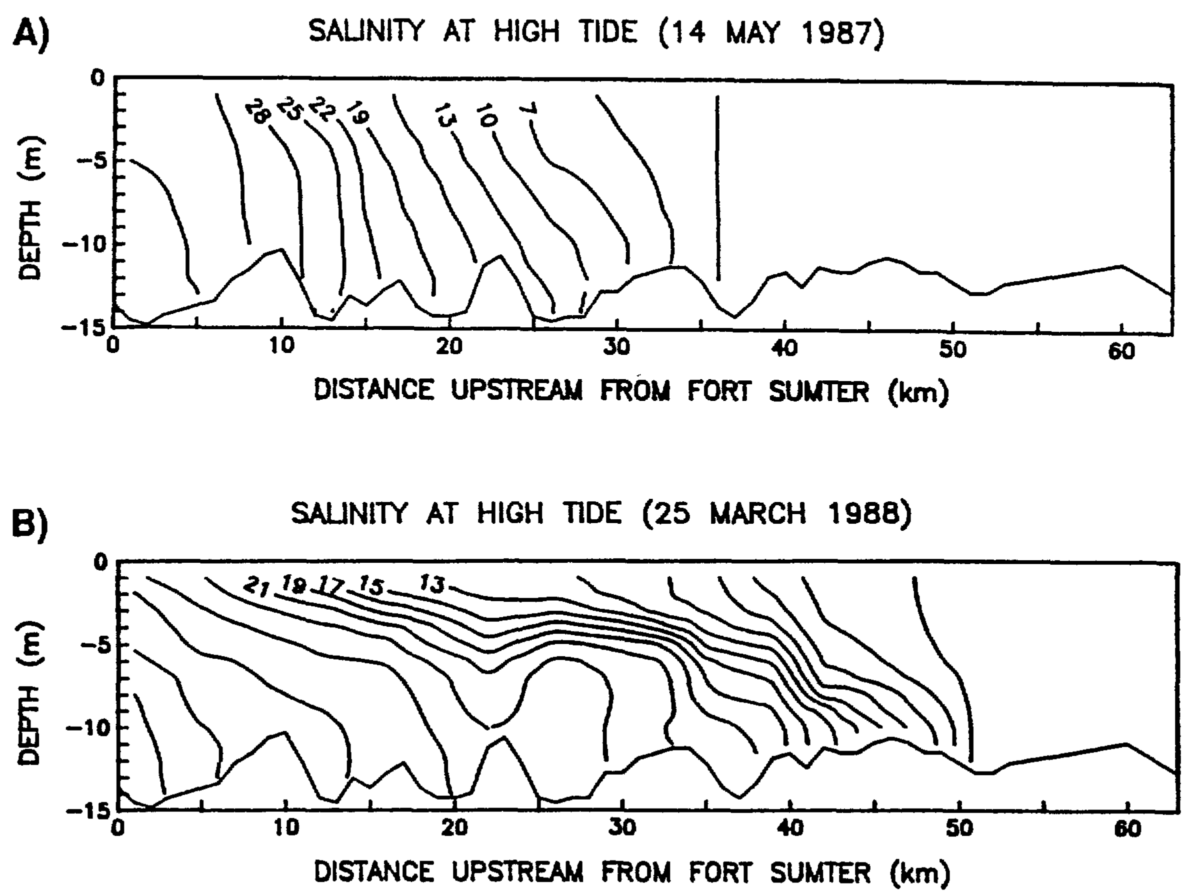

Figure V.6. High tide salinity distribution along the Cooper River from Fort Sumter (0 km) during (A) a spring tide and (B) a neap tide.

data, examples shown in Figures V.7 and V.8, indicate pronounced vertical as well as lateral structures in both instantaneous and net (not shown) salinity and velocity distributions, suggesting the importance of both gravitational circulation and residual tidal circulation at the entrance to Charleston Harbor. The transmissivity/TSS data (not shown) show similarly strong lateral gradients across the entrance. As a next step, salt and sediment fluxes will be decomposed (e.g., Bowden, 1963; Hansen, 1965; Kjerfve, 1986) and analyzed for the mechanisms responsible for the majority of the fluxes.

The implementation of the numerical model for Charleston Harbor has progressed in a series of steps. The initial choice of 500 m grid cells was abandoned in favor of 300 m grid cells to improve on the spatial resolution. All initial runs were made assuming that the salt marsh remained dry at all stages of the tide. However, this was not satisfactory, as the simulated tidal prism only measured 50% of the actual tidal prism. Thus, it became necessary to introduce the moving model boundary to allow for the marsh elements to become flooded as the tide exceeds the marsh surface elevation. As a consequence, simulated and actual tidal prisms now agree closely, and the dynamics of the modeled system behaves differently as compared to the fixed boundary grid situation. Much of the recent model work has been focused on a series of verification runs with Manning's n

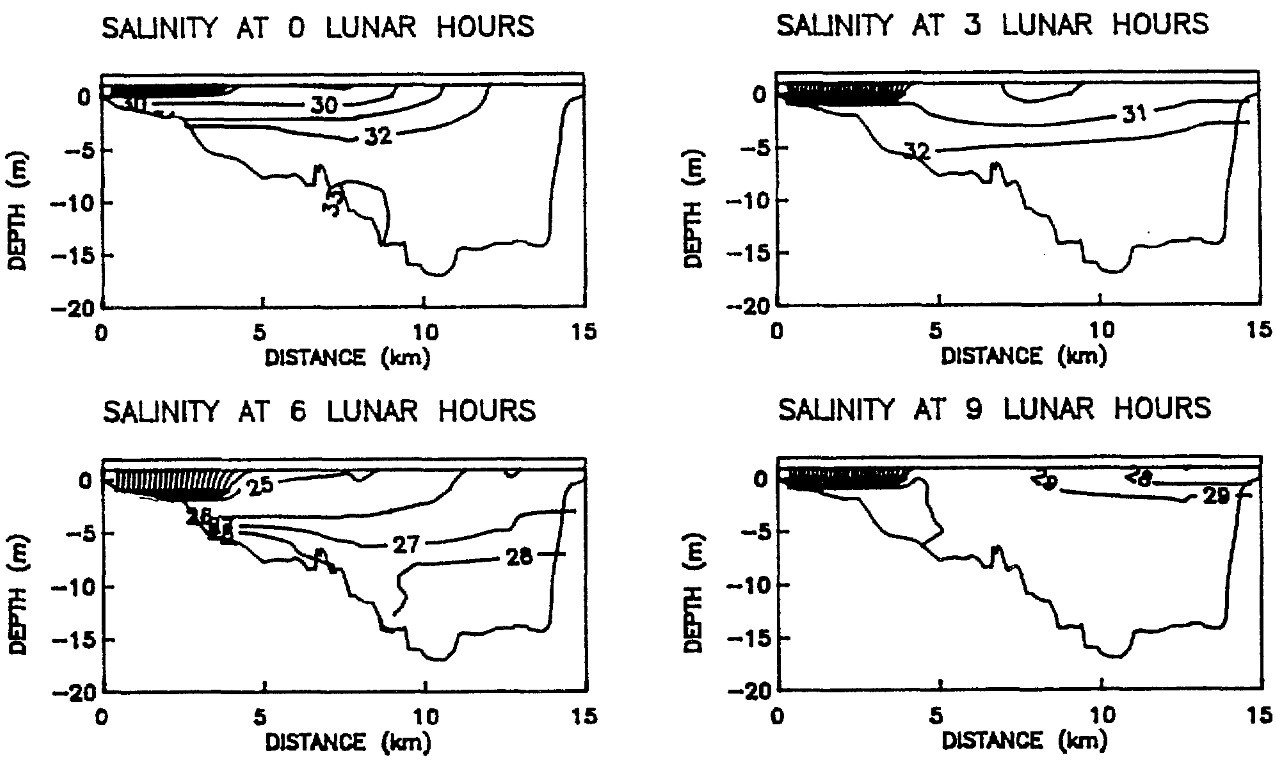

Figure V.7. Instantaneous cross-sectional plots of salinity in the Fort Sumter to Fort Moultrie cross-section, shown every 3 lunar hours over a neap tidal cycle, 18 February 1988 (JD 49).

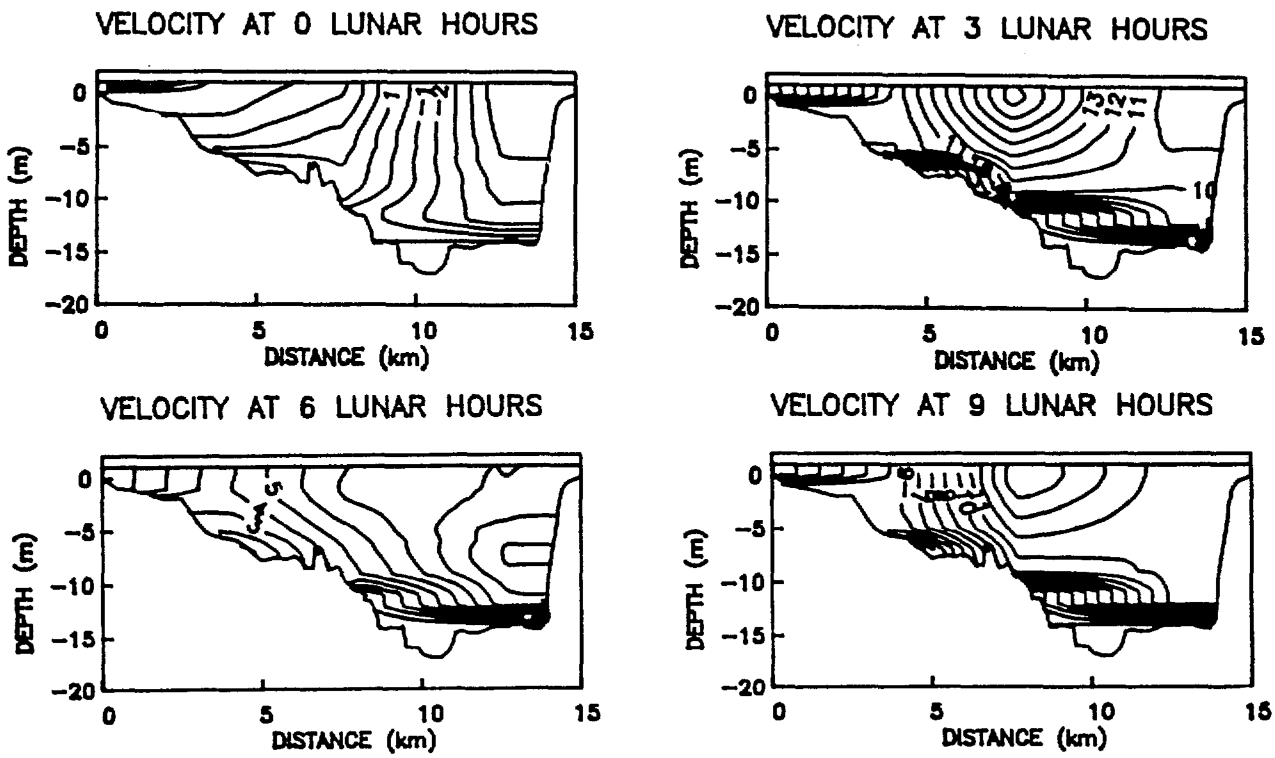

Figure V.8. Instantaneous cross-sectional plots of velocity in the Fort Sumter to Fort Moultrie cross-section, shown every 3 lunar hours over a neap tide cycle, 18 February 1988 (JD 49).

varying from 0.01 to 0.05 for the estuarine channel elements and from 0.10 to 0.30 for the occasionally flooded marsh elements. Thus, the actual task of simulating numerical case studies has just recently begun, and all results up until now are of a preliminary nature.

The complete grid for the Charleston Harbor system, including the Cooper River model, is shown in Figure V.9. Verification and preliminary model runs have been executed with 126 m^3/s freshwater discharge from the Cooper River, no winds, and forcing by an M_2 tidal wave with an amplitude of 0.9 m at the harbor entrance. To satisfy the Courant-Levy-Fredericks (CLF) computational stability criterion, we selected a time step of 6 s. The hydrodynamic portion of the model generally reaches quasi-steady state after 3 tidal cycles, whereas the dispersion portion of the model only reaches quasi-steady state after 20-25 cycles.

Model outputs include numerical values as well as graphical distribution plots for any desired time-step. Computed parameters include velocity, water elevation, and salinity. After achieving quasi-steady state, the outputs for a complete tidal cycle are averaged to yield: (1) transport velocity vectors; (2) Eulerian velocity vectors; (3) Stokes' velocity vectors; (4) net salinity; and (5) net water elevation. Because the tides cause the local

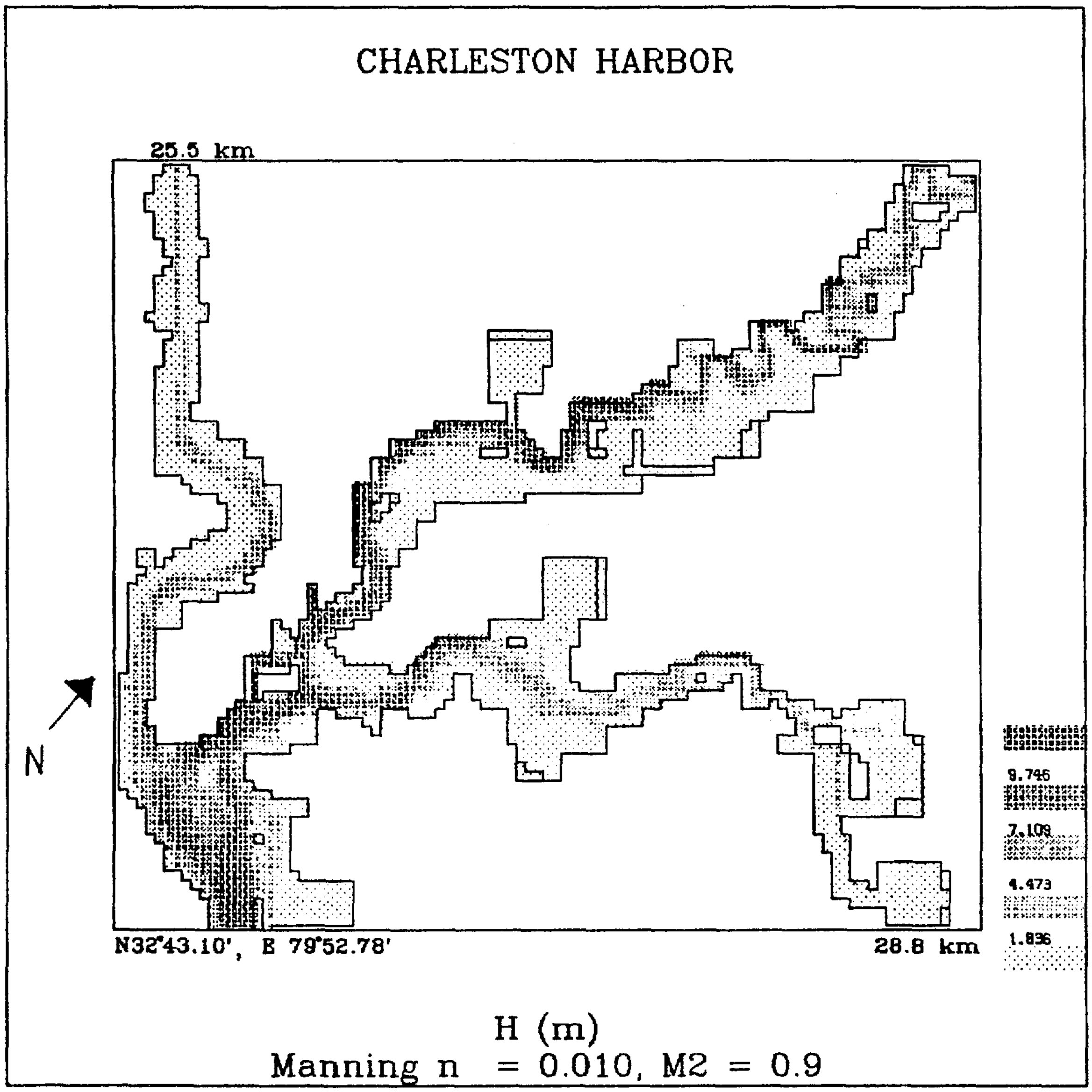

Figure V.9. The numerical domain for Charleston Harbor and the Cooper, Ashley, and Wando Rivers, with $\Delta y = \Delta y = 300$ m, showing model bathymetry and the occasionally flooded marsh elements.

water depth to change significantly with time, it is necessary to calculate both the Eulerian (usually seaward) and Stokes' (usually landward) velocities and the combined transport velocity (Cheng and Casulli, 1982). Whereas the Eulerian velocity does not account for the tidal wave transport over a tidal cycle, the Stokes' velocity is a measure of the residual transport due to the tidal wave. The combination is the transport velocity, which resembles the Lagrangian flow (Cheng, 1988). For Charleston Harbor, the landward Stokes' transport is usually one order of magnitude smaller than the seaward Eulerian transport, but does become dominant in shallow areas of the estuary, where the tidal range accounts for most of the total water depth.

As an example of model output, the net transport velocity field is shown (Figure V.10) for a simulation of forcing by an M_2 tide of 0.9 m amplitude, Cooper River discharge of 125 m^3/s, no winds, and a Manning's n of 0.01 in the harbor proper, then decreasing down to 0.05 in the harbor entrance channel. The simulation was performed over the entire linked two-dimensional/one-dimensional model domain, but for better visual

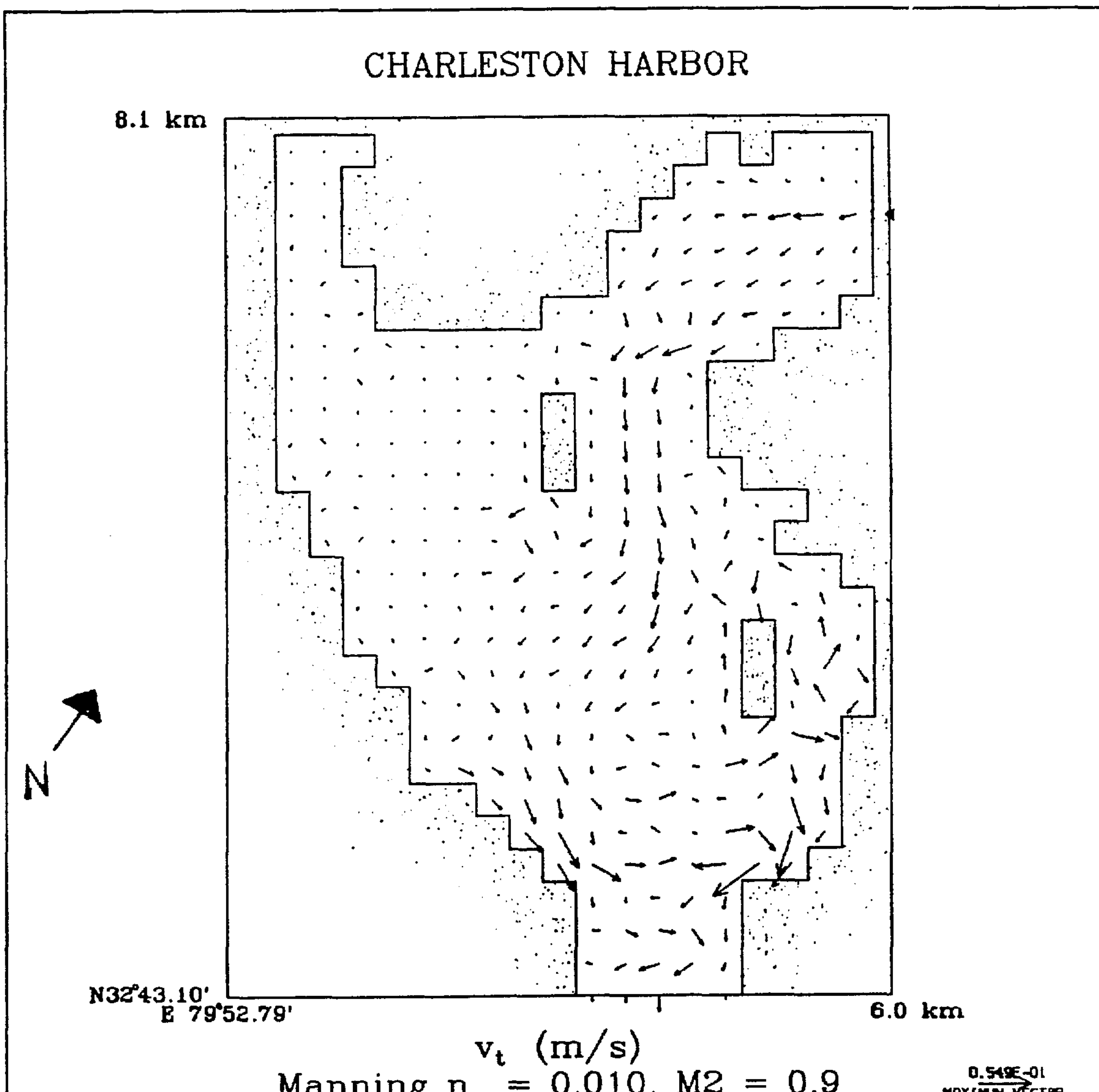

Figure V.10. Example of model output, showing the net transport velocity over one tidal cycle. The model was forced by an M_2 tidal wave of 0.9 m amplitude, Cooper River discharge of 125 m^3 /s, no winds, and a Manning's of 0.01 in the harbor decreasing to 0.05 in the entrance channel.

interpretation, we show here only an enlarged view of the harbor proper. Net velocities during this simulated tidal cycle average 5 cm/s. The figure shows a net transport out of the harbor due to freshwater discharge from the Cooper River. Large down-estuary flow occurs along the Cooper River channel, then slows down, deflects to the left, and flows south along the Ashley Channel, forming an eddy-feature to the left of the lower island in the grid (Crab Bank). This flow pattern is probably the result of bathymetry changes as well as the presence of Crab Bank island, which causes currents to deflect around it, forming small eddies.

SUMMARY

1. The Charleston Harbor estuarine system has undergone major hydrological changes as a result of diversion of a portion of the Santee River flow into the Cooper River in 1942, and then rediversion of the Cooper River flow in 1985, thereby reducing its mean discharge from 425 m^3/s to 122 m^3/s. The objective of this study was to assess estuarine salinity responses to the freshwater flow rediversion and to develop the capability to model and predict physical dynamic processes in the Charleston Harbor estuary.

2. Intensive hydrographic sampling was conducted in Charleston Harbor and the Cooper River from 1987-1988. Measurements obtained include current velocity records and vertical profiles of conductivity, temperature, density, and transmissivity, collected along a longitudinal transect of the Cooper River and a cross-sectional transect of the Harbor mouth. A two-dimensional numerical circulation/dispersion model of the Charleston estuary was developed and extended to include tidal portions of the Ashley, Cooper, and Wando rivers. The numerical model, together with the hydrographic field data and additional data obtained from USGS and NOS/NOAA, are being used to assess salinity responses of the Charleston Harbor/Cooper River estuarine system.

CHAPTER VI

SEDIMENTS

by

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INTRODUCTION

Prior to 1942, natural conditions in Charleston Harbor were such that maintenance dredging in order to provide a 30 foot navigational channel to the Port Terminal was insignificant (Simmons, 1972), with the gross annual maintenance dredging in the harbor channels averaging about 16,5649 m^3 (21,7000 yd^3) (Mathews *et al.*, 1980). In the early 1940's, deepening of navigation channels to 35 ft and diversion of the Santee River into the Cooper River combined to increase sedimentation in the harbor significantly. The Cooper River, once a drain for 3077 km^2 (1188 mi^2) of the Coastal Plain of South Carolina, suddenly became the major discharge route for 40,674 km^2 (15,700 mi^2) of watershed (Neiheisel and Weaver, 1967). The ensuing increase in suspended sediment from upland sources, scour of bed and banks, and disruption of the estuarine hydrography resulted in an average increase in volume of dredged material to 2.9 million m^3 (3.8 million yd^3) per year (USACOE, 1966). Further alterations of navigational channels prior to rediversion compounded sediment deposition problems to the point where 5.8 million m^3 (7.6 million yd^3) of sediment had to be removed from the inner channels of Charleston Harbor in order to maintain required depths in 1982 (Patterson, 1983).

Sedimentation in Charleston Harbor is derived from both marine and fluvial sources. Deposition is influenced by freshwater inflow and modifications of bottom topography. As described in detail in the hydrographic section of this report, the Charleston Harbor estuary changed from a well-mixed to a partially stratified system with the increased flow resulting from diversion of the Santee River (Simmons, 1966; USACOE, 1966; Meade, 1969). The corresponding saltwater wedge is primarily responsible for sedimentation in the lower Charleston Harbor estuary. Sediment transported along the harbor bottom encounters a point of no-net motion at the saltwater/freshwater interface. Over the course of a given tidal cycle, the advance and retreat of the saltwater wedge creates points of no-net motion (*i.e.* nodal points) at which deposition most readily occurs. Sediment entrained within the saltwater wedge moves upstream and downstream, settling where current is insufficient for transport, only to be resuspended and redeposited in following tidal cycles (Neiheisel and Weaver, 1967; Van Nieuwenhuise, 1978).

A high proportion of the bed material in this region has been deposited as a result of flocculation (Neiheisel and Weaver, 1967). The cohesion associated with this process forces clay particles and colloids to aggregate and settle in a manner similar to that of larger particles. While transported within a freshwater medium, clay particles maintain a negative charge. However, upon encountering salt water, the charge becomes neutral, promoting an attraction between particles. Moderate turbulence or shear rate adds to this chemical reaction by further inducing flocculation. Consequently, high concentrations of suspended solids and moderate turbulence result in flocculation as salt water is encountered, with tides and freshwater discharge being the determining factors of actual dispersal (USACOE, 1966). Prior to rediversion, dispersal tended to be towards the western side of the Charleston Harbor estuary, extending up the Cooper River to the Navy Yard (Neiheisel and Weaver, 1967).

The source of the sediments dispersed by these phenomena was disputed. Simmons (1966) suggested that fluvial, upstream sediment was trapped within the saltwater wedge when predominantly landward-moving bottom currents met with the seaward-directed fresh water. Meade (1969) contended that the landward flow was responsible for transporting sediment from longshore and offshore sources into the estuary where deposition occurred. Neiheisel and Weaver (1967) used ratios of different clay minerals in bottom and suspended sediments as diagnostic indicators of sediment origin. Relative proportions of clay minerals known to originate in separate geological provinces (Piedmont, Coastal Plain and Continental Shelf) were used in conjunction with salinity and velocity data to ascertain source and dispersion patterns. Analyses of both bottom and suspended sediment samples revealed that sediments transported down the Cooper River consist primarily of silt and clay. The ratios of the clay constituents indicated that the sediment originated primarily in the Piedmont and that it was settling in proximity to the western shoals of the harbor basin. In addition, flow studies indicated that freshwater discharge from the Cooper River was directed toward the western portion of the estuary. Sand-size particles in the harbor were found to be transported by littoral drift from the beach and around Sullivans Island spit. Deposition occurred at regions of insufficient energy within the estuary.

Van Niewenhuise, *et al.* (1978) used another tracing technique, Fourier Series grain shape analysis, to draw similar conclusions. Sand and silt fractions were separated, size frequency distributions were calculated and distribution patterns plotted. It was found that sand originating from the proximal continental shelf is the major cause of shoaling in the navigational channels of Charleston Harbor. Marine sands dominated the sediment in the channel corridor from the harbor entrance to about 18 km up the Cooper River, thus supporting Meade's theories of landward transport to the extreme nodal point. Grain-shape analysis of the silt fraction indicated that it was predominantly fluvial and that it was distributed by net seaward flow of the surface layer. In fact, distribution of the river-derived silt reached a seaward extreme that also marked the landward extreme of marine sands.

For comparative purposes, the investigation of local stratigraphy and sedimentation in the Charleston Harbor area conducted by Colquhoun (1972) is the most thorough assessment of sediment distribution during the diversion period from 1942-1985. Through the use of seismic profiling and bottom sample analyses, surficial sediments were described for the harbor, adjacent rivers and the immediate continental shelf. Sands were found to dominate the proximal shelf region extending throughout the northern and central areas of the harbor basin. Sand was also the most prevalent bed material in the Wando River and upper Cooper River. Silt and clay characterized sediment type within the Ashley River, south and west sides of the harbor and extensive portions of the lower Cooper River. In addition, organic content of bottom sediments was found to be most common "at or seaward of high human activities."

In 1966, the USACOE concluded that rediversion of the Cooper River water flow back into the Santee River system would significantly curtail landward bottom currents responsible for sediment trapping and affect the distribution of various sediment types within the estuary. Shoaling was estimated to diminish by 70% (USACOE, 1966).

Sedimentological sampling was conducted in the present study to:

1) Compare sediment characteristics at several index sites over a four-year period, which encompassed rediversion, in conjunction with macrobenthic infaunal sampling.

2) Describe the spatial distribution of surficial sediments within the harbor basin and lower estuarine reaches of the Ashley, Cooper and Wando Rivers based upon a more spatially intensive sampling design also in conjunction with infaunal sampling.

3) Compare sedimentological data from these studies with data from studies conducted prior to rediversion.

METHODS

Bottom sediments for the long-term seasonal study were quantitatively sampled at thirteen sites located throughout the Charleston Harbor estuary (Figure VI.1). Sampling in the Cooper River, Wando River and harbor basin was conducted quarterly over a four-year period in order to detect seasonal and annual fluctuations. Sampling of the Ashley River was conducted during a one-year period commencing in November 1987. Site locations were selected to span most of the estuarine gradient as well as to provide comparative data with pre-rediversion studies in this area. All stations were initially located relative to fixed landmarks and supported later by Loran-C positioning. Three replicate 0.05-m^2 grab samples were collected seasonally at each site. A vertical cross section of approximately 100 ml of sediment was subsampled from one of the three replicates in order to determine sediment character associated with the infaunal communities.

Sediments for the short-term intensive study of benthic macrofauna were sampled at 178 stations located throughout the Charleston Harbor estuary in the summer of 1988 (Figure VI.2). Stations were located equidistantly along transects that were perpendicular to the maintained navigational channels and separated by a distance of approximately 1 km. The three-meter depth zone marked the shoreward extent of each transect. Loran-C coordinates were plotted for most stations prior to the field effort to assist in station location. Unlike the seasonal study, only one 0.05-m^2 grab sample was taken at each site. One core of approximately 100 ml was extracted from this sample for analyses of the ambient sediment. Cross sections from the long-term seasonal study and cores from the short-term intensive survey were comprised of sediments from surface to maximum depth of penetration of the grab. Both were prepared and analyzed identically. In the laboratory, sediment subsamples were air dried, then manually disaggregated and split into two portions. Three grams were randomly separated from the subsample for the analysis of organic matter and the remainder subjected to further mineralogical and textural analyses.

Percent weight of organic matter was determined by drying samples completely in a desiccator oven for eight hours at 100°C, then placing them into a muffle furnace for two hours at 550°C to eliminate the organic carbon component (Plumb, 1981). Percent weight of organic matter was then calculated for each sample.

Textural analyses were performed by placing each sample in a dispersing solution and disaggregating for fifteen minutes with a sonic dismembrator. Distilled water was then added to provide a 1000-ml solution for standard hydrometer analyses of sand, silt and clay. Material smaller than 62.5 microns was sieved and discarded. The remainder was

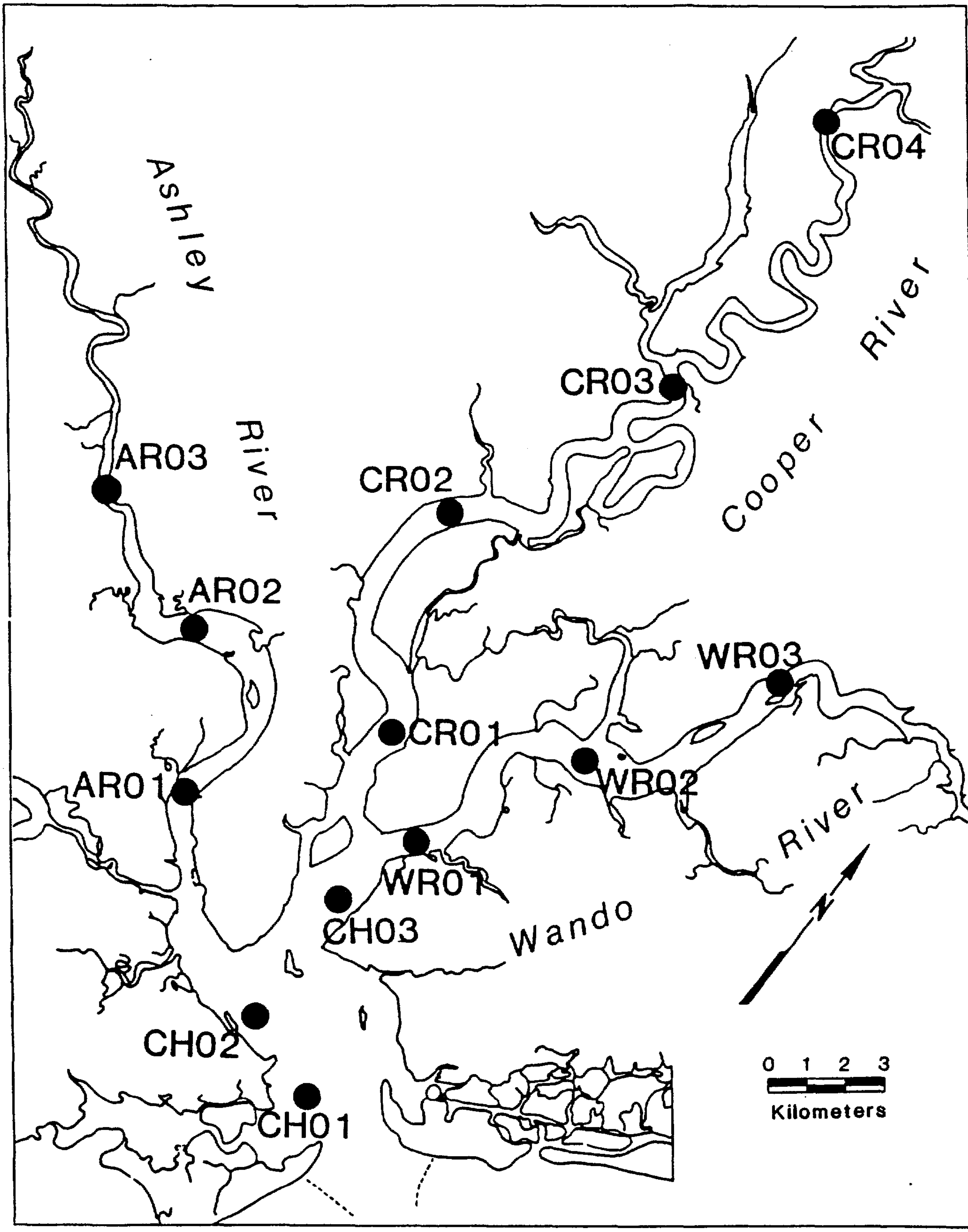

Figure VI.1. Approximate location of stations sampled seasonally for benthic communities and granulometry.

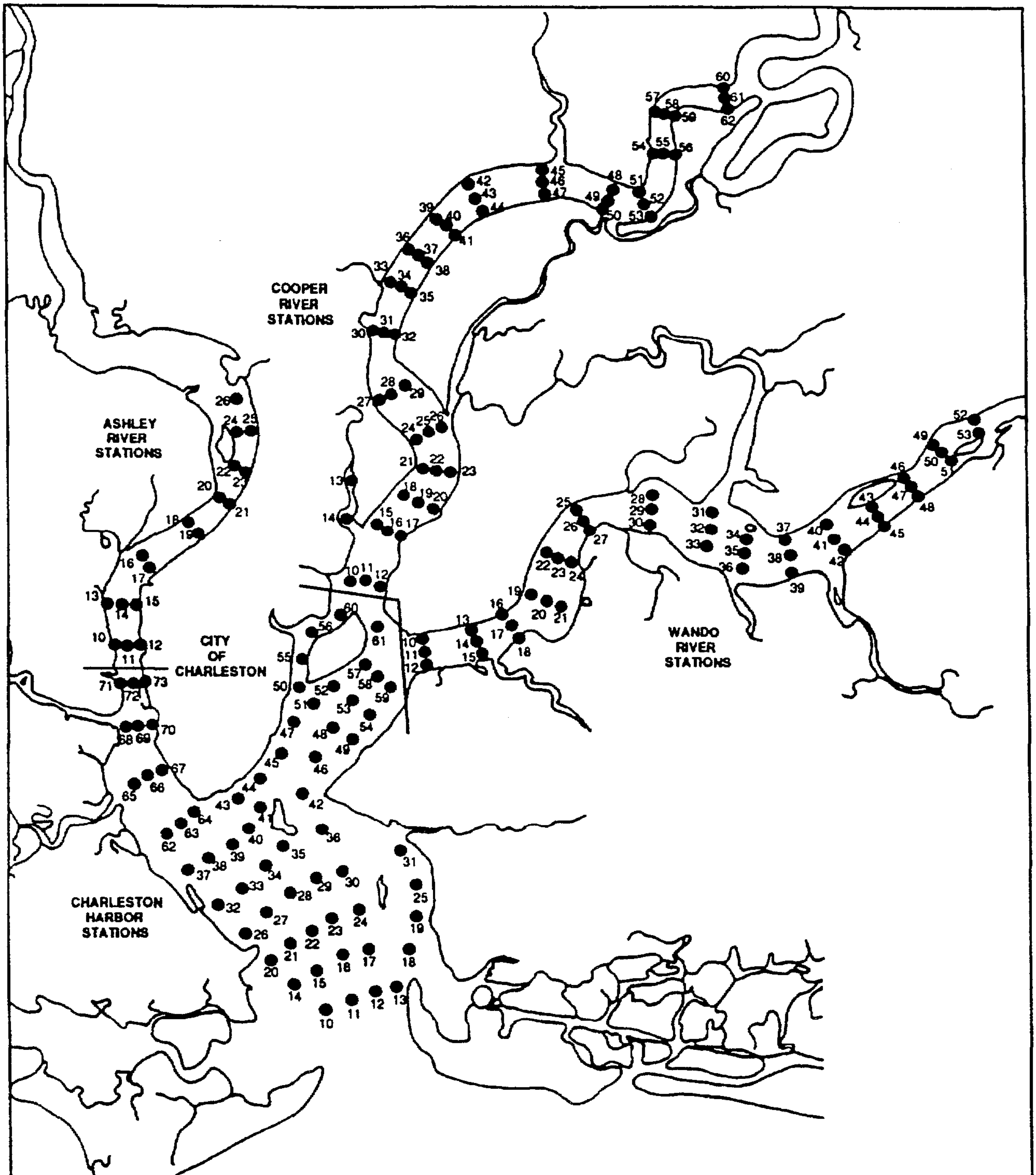

Figure VI.2 Approximate location of stations sampled for the special benthic study.

then dried at 100°C for eight hours. A mineralogical analysis was then performed to determine percent weight of calcium carbonate. Dilute (10%) hydrochloric acid was added to each sample, until effervescence ceased, to dissolve the carbonate fraction. All samples were then incinerated for two hours at 700°C in order to eliminate any remaining organic matter prior to the sand grain size analysis. This was performed by dry-sieving each sample for fifteen minutes using a mechanical shaker fitted with fourteen sieves graded in 0.5-phi intervals (Pequegnat, 1981). Individual and cumulative weight percentages were determined for each size class and used to generate a cumulative frequency curve for each sample. Mean grain size, sorting, skewness and kurtosis were hence derived from these frequency distributions.

RESULTS AND DISCUSSION

Seasonal Study:

Charleston Harbor - The three stations located within the harbor basin displayed the greatest variability in sediment composition between stations as well as over time (Figures VI.3 and VI.4). Sediments from stations CH01 and CH02, both located along the southern portion of Charleston Harbor, displayed no consistent seasonal patterns and did not relate to the commencement of rediversion (Figure VI.4). The high variability that was evident prior to rediversion continued after water flow became more constant. Sand, silt and clay were the most variable; however, the lesser components of organic matter and calcium carbonate or shell hash were also highly variable at CH02 (Appendix VI.1). Although fluctuations in the percentage of organic matter coincided with those of silt over the course of the study, this is not necessarily indicative of particle size. Organic matter deposition is closely related to density, as well (M. Katuna, pers. comm.). It is not known to what degree estuarine organic matter originates from anthropogenic sources as opposed to detrital decomposition in this system; however, it was frequently associated with man-made structures. Concentrations of calcium carbonate were highest at CH02, ranging from 1% to 47%. The dwarf surf clam, *Mulinia lateralis,* occurred sporadically in elevated numbers at this station, and shells from this species may have influenced the calcium carbonate content of the sediment samples. However, no obvious relationship existed between calcium carbonate content and *M. lateralis* abundances at this station. Levinton (1970) found that high juvenile mortality of *M. lateralis* can result in a numerically dominant occurrence of this species within the fossil assemblage of muddy substrates although it is often a lesser component of the living population (see Chapter VII). The temporal record of the sand constituent at CH02 was far more erratic than at the other two harbor stations as well, regardless of water flow.

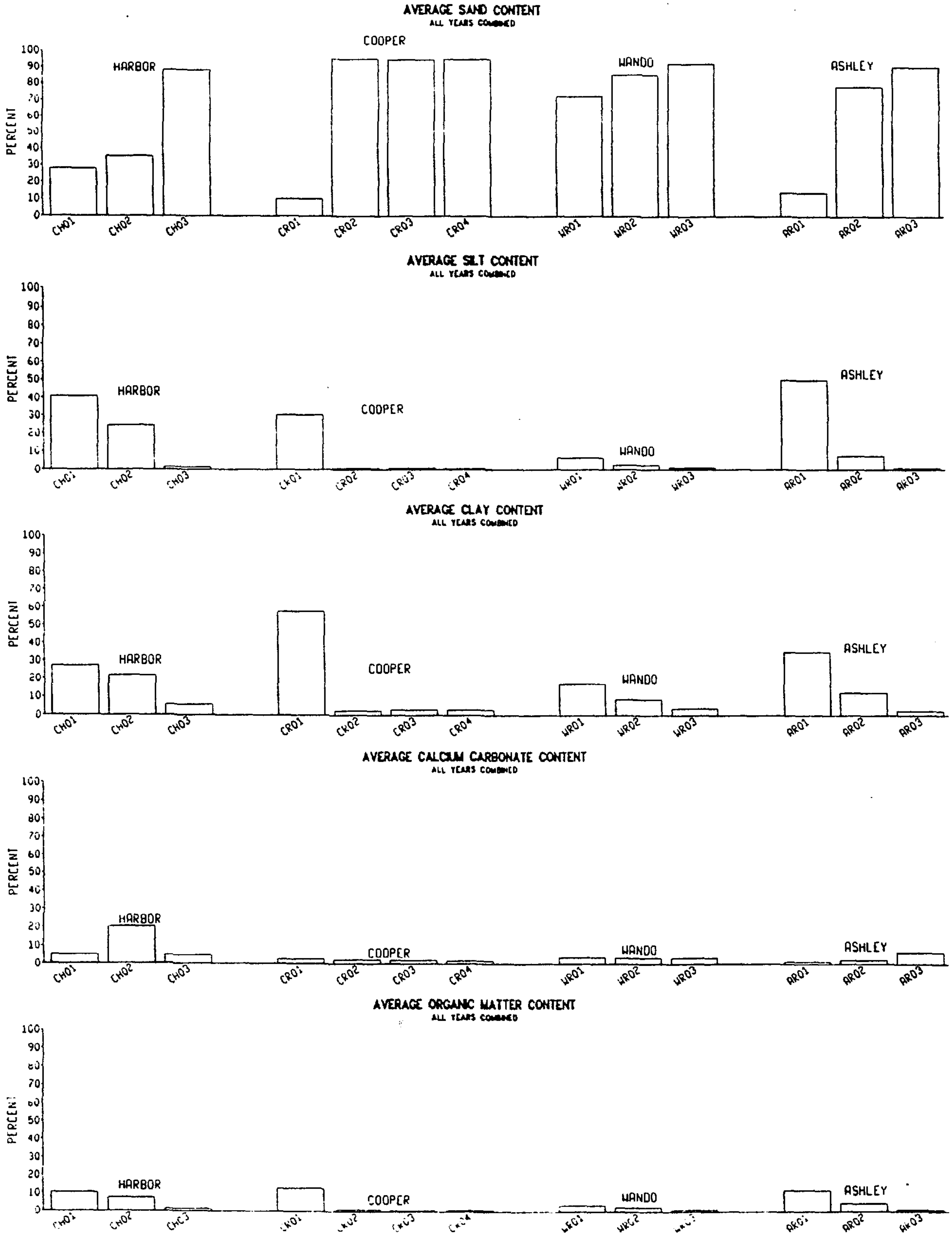

Figure VI.3. Average occurrence of sedimentological components analyzed.

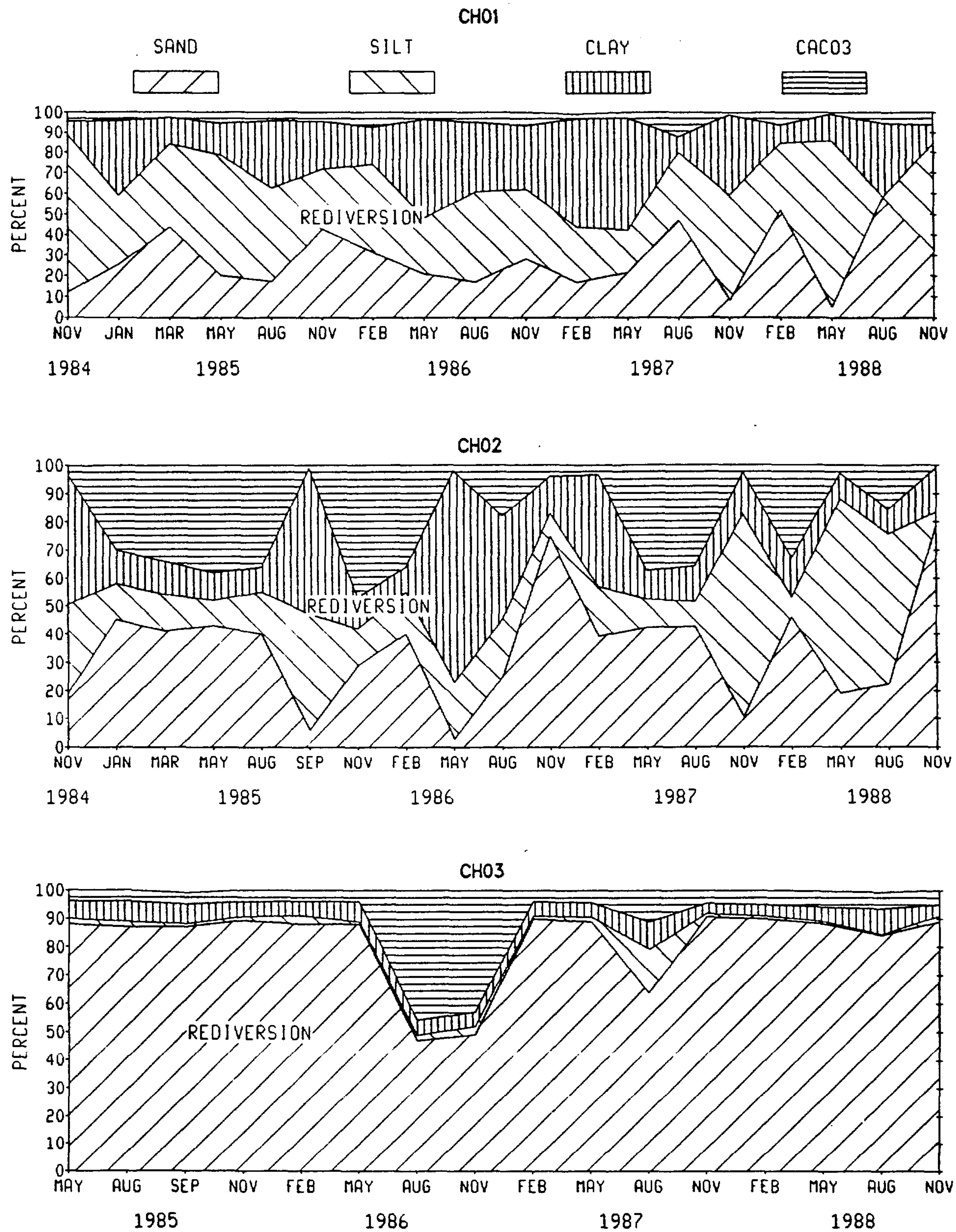

Figure VI.4. Percentages of sand, silt, clay and calcium carbonate at each of the Charleston Harbor sites.

Sediments at station CH03 were less variable in composition than those at the other two sites. Aside from three aberrant samples, sediment at this station was consistently composed of well-sorted, fine sand (Figure VI.4). The inconsistencies noted in the data for August and November of 1986 and August of 1987, were most likely due to imprecise site relocation; therefore, these data were omitted from any data analyses of this station.

All three harbor stations are located on gradually sloping flats adjacent to maintained channels (Figure VI.1); however, CH03 is more directly aligned with discharge from the Cooper and Wando Rivers. Sorting and transport of fine bed material may be facilitated by the higher energy of this region. Proximity to dredging activities within the navigational channels may also contribute to the variability witnessed at CH01 and CH02. Mixing and resuspension of bottom sediments occurs as part of the dredging process and may have an effect on bed material for an extensive area. Imprecise station relocation may also account for the substantial fluctuations in sediment composition on an occasional basis. Still, when located on sediment distribution plots for pre-rediversion (Colquhoun, 1972) and post-rediversion, as generated by the intensive survey conducted in July 1988, it is apparent that overall sediment types are consistent for these three stations.

The net effects of rediversion upon sedimentation are not expected to be evident for approximately ten years after its commencement (Patterson, 1983; Teeter, 1989). Teeter's (1989) predicted reduction in overall shoaling of 74% at 3,000 cfs recognizes that the current deepening project, shifts in unconsolidated mud, and natural variability in the hydrodynamic regime will delay conclusions.

Cooper River - The four stations located in the Cooper River displayed a more pronounced gradient relative to river mile (Figures VI.3 and VI.5). Station CR01, nearest the harbor (Figure VI.1), had the greatest percentage of fine material, although it fluctuated substantially throughout the study. The three upriver stations all had medium to well-sorted, medium sand, whereas CR01 sediments were comprised primarily of silt and clay (Figure VI.5). Exceptions to this pattern were noted during the February 1988 sampling for CR01 and November 1985 for CR04. Lack of a consistent change indicates that these data were probably influenced by errors in station location; hence, these particular data were not included in subsequent interpretations and are presented only for graphic comparisons.

Seasonal trends were not apparent at any of the four Cooper River stations. Station CR01 displayed erratic changes in surficial sediments from one sampling effort to the next. The other three stations remained consistent over time regardless of season or year (Figure VI.5).

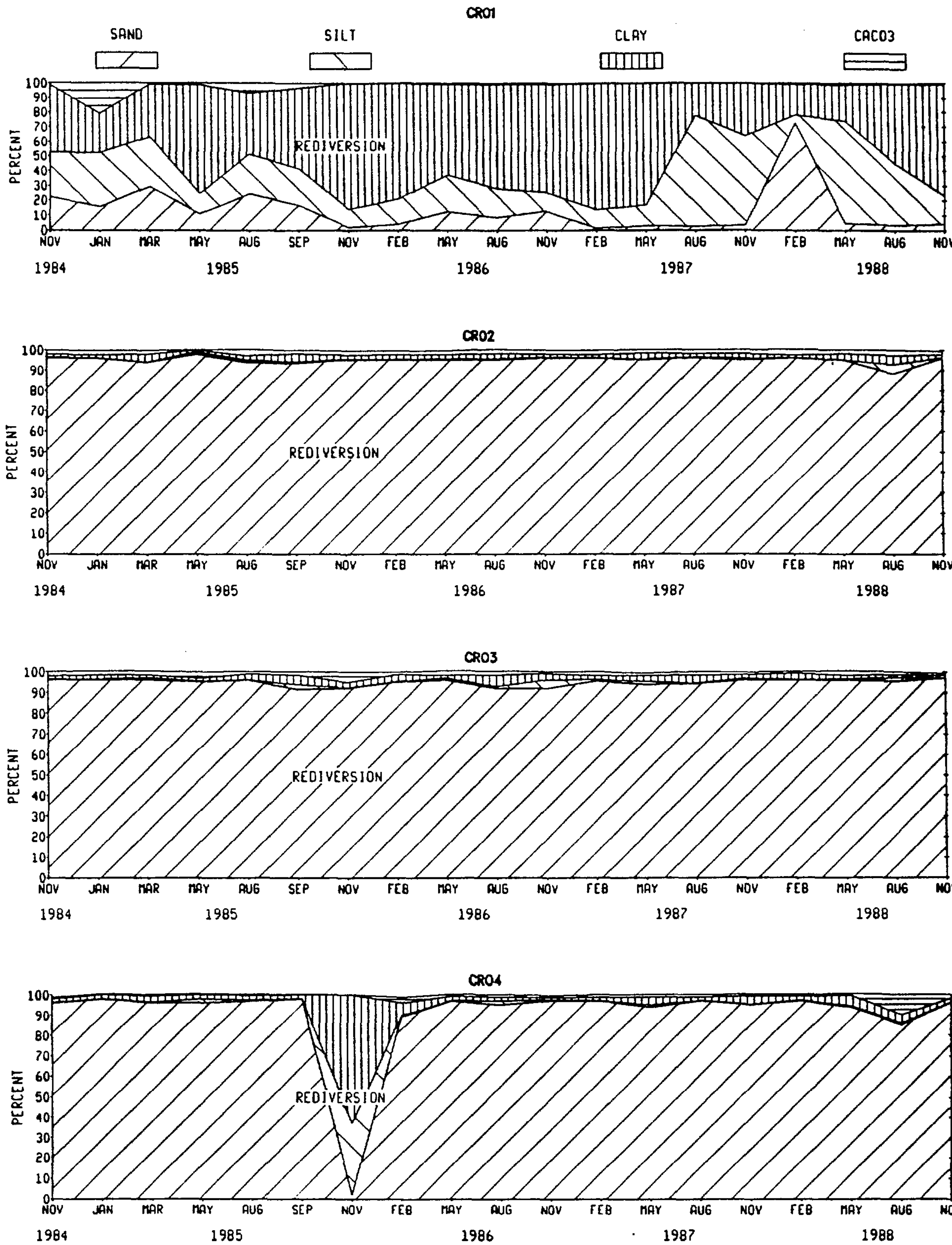

Figure VI.5. Percentages of sand, silt, clay and calcium carbonate at each of the Cooper River sites.

There were no obvious effects of rediversion upon sediments at most of the Cooper River stations. While the stratified tidal fronts have shifted upriver (see Chapter III), there has not been an obvious coinciding shift in the occurrence of silt and clay associated with this phenomenon (Simmons, 1966; Van Nieuwenhuise, 1978; Simmons and Herrmann, 1972). Subtle fluctuations in surficial sediment characteristics were very likely obscured by the need to sample the full extent of the penetration of the grab. The lower three stations have been only partially mixed at high slack current since rediversion while, prior to this event the nodal point for the density current was situated 8 to 11 miles upriver (Simmons, 1966), encompassing the region between CR01 and CR02 (Figure VI.1). Still, over the course of the three years following rediversion, sediment at station CR01 had an average of 20% more silt and clay (by weight) than the year prior to rediversion. This occurred as an immediate increase over the first two seasons following reduced fresh water discharge (Figure VI.5). Subsequent field efforts revealed fluctuations in the silt/clay fraction; however, it remained elevated above pre-rediversion conditions.

Sand sorting coefficients for CR04 may have been influenced by decreased water flow following rediversion, also. Prior to August 1985, surficial sand at CR04 was very well sorted (Appendix VI.1). After rediversion, however, the sand fraction ranged from very poorly to very well sorted. Sediments at the three stations downriver displayed more consistent sorting coefficients throughout the course of this study.

The mineralogical components of the Cooper River stations sampled quarterly over four years indicated no substantial changes related to season or rediversion at the three upriver stations. At CR01, however, the percentage of calcium carbonate diminished immediately upon commencement of rediversion and continued to represent less than one percent by weight of the total sample (Appendix VI.1). This was similar to the trend exhibited by the sand component, suggesting that the coarser sand and shell hash material were gradually representing less of the total weight of each sample as the occurrence of silt and clay increased. Organic matter was a major component of the sediment at CR01 in relation to the other three stations in this river. No seasonal trends or effects of rediversion were observed, and the relatively high percentage of organic material at this site may be due to its specific topography and hydrography (see Chapter III).

Wando River - The Wando River was less directly affected by rediversion; however, its hydrography is greatly influenced by the Cooper River as a result of mixing at the confluence of these two rivers and from Beresford Creek (SCWRC, 1979). Based upon cross sections collected in the Wando River, bed material sampled at the three Wando River stations showed little indication of either an immediate shift in composition or a consistent pattern of change during the three years following rediversion (Figure VI.6). Station WR01 displayed an overall shift towards a sandier substrate; however, strong non-seasonal fluctuations throughout the four-year span of this study hinder interpretation or prediction of the textural components at this station. Consistent with upriver stations in the Cooper River, WR03 sediments were composed primarily of fine to medium sand prior to rediversion and remained as such for the duration of this study (Appendix VI.1). Though WR03 displayed none of the abrupt changes in sediment type noted at the sites downriver, there still remained no indication of seasonality (Figures VI.3 and VI.6).

The calcium carbonate content at all stations in the Wando River averaged approximately 3-4%, displaying no apparent seasonality or effects of rediversion (Appendix VI.1). Organic matter was also a relatively minor component of the sediments in this river (1-3%). However, among these three sites, the occurrence of organic matter diminished with increasing distance upriver.

Ashley River - Ashley River stations (Figure VI.1) were sampled during five seasons beginning in November of 1987. Thus, there are no pre-rediversion data for these specific locations. However, Colquhoun (1972) provided some limited information regarding bed material in the Ashley River based upon seismic profiles and borehole data. General areal distributions of sediment types were derived from these data while recognizing the limitations to specific locations or detection of subtle changes.

The Ashley River was not directly affected by rediversion above its confluence with the harbor basin, and this system is not connected to the Cooper River at any point (Figure VI.1). Tidal fluctuations dominate the hydrography of the Ashley River which has relatively little freshwater input compared to the Cooper River. As a result, stratification at the three Ashley River stations during high slack current was minimal, with an average of approximately 1.5 ppt difference in salinity from surface to bottom after rediversion. Sediment deposition was probably influenced accordingly, with respect to silt and clay (Simmons, 1966).

Surficial sediments along several transects in the Ashley River, as analyzed by seismic profile (Colquhoun, 1972) were dominated by silt and clay. Transects were confined primarily to the channel, extending from the mouth to just above the Highway 17 bridge. This sediment type is consistent with that found in post-rediversion sampling at station AR01, only. If bed material in the Ashley River had been influenced by rediversion and the associated shifts in hydrographic regime, one might expect it to have been most obvious in the lower reach of the river which encompassed this station. Stations AR02 and AR03

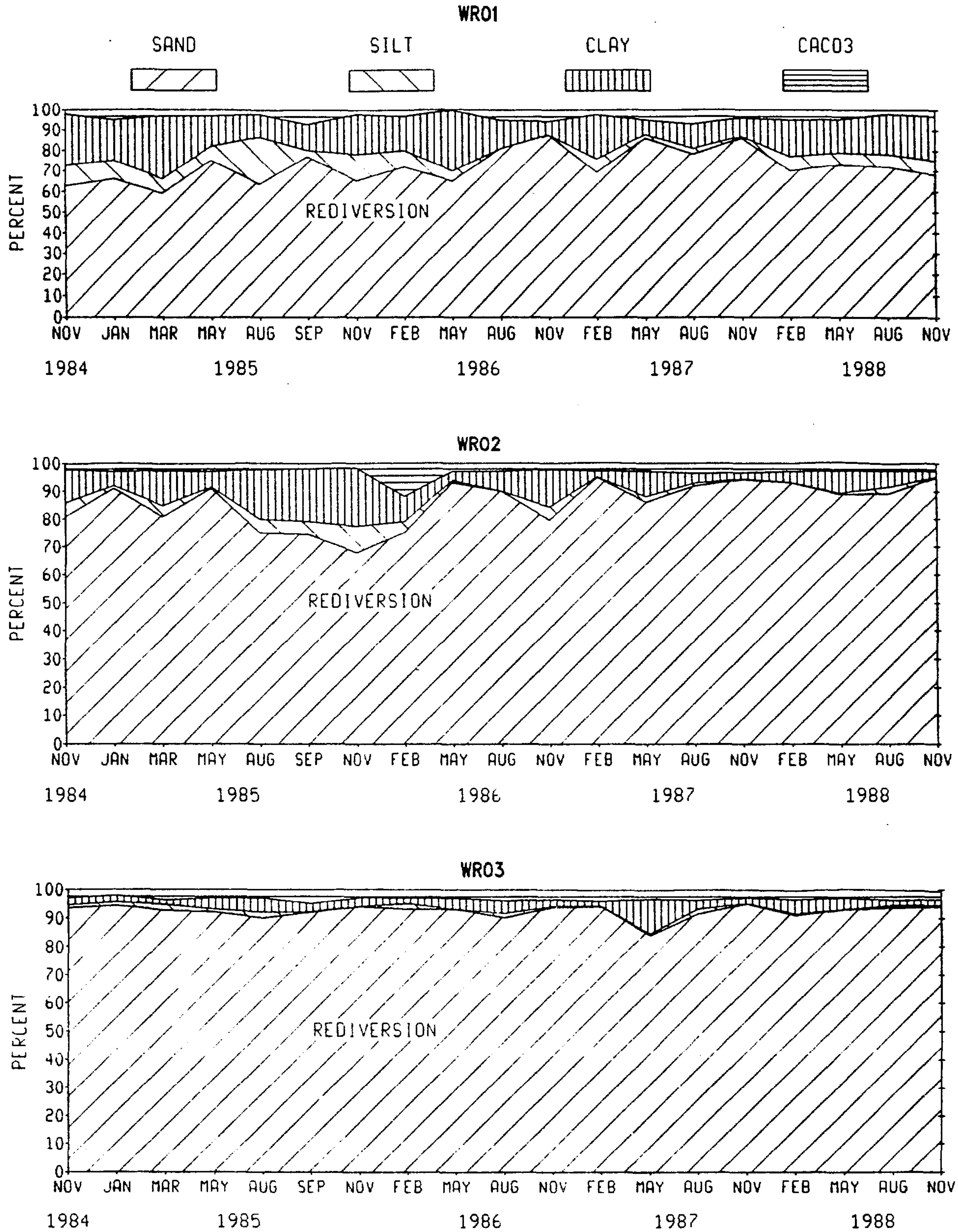

Figure VI.6. Percentages of sand, silt, clay and calcium carbonate at each of the Wando River sites.

both lie beyond the extent of the seismic profiling tracks and had sediment that differed from those described by Colquhoun. Given the limited database prior to rediversion and the short duration of post-rediversion sampling, little can be concluded regarding changes related to this event or trends since that time (Figure VI.7).

As in the Cooper and Wando Rivers, stations in the Ashley River were sandier proceeding upriver (Figure VI.1). Ashley River stations also displayed greater temporal consistency with increasing river mile. In addition, mean grain size of the sand fraction was coarser upriver, grading from fine at AR01 to medium and coarse at AR03 (Appendix VI.1).

The occurrence of calcium carbonate at the three Ashley River sites increased upriver. Conversely, organic content reflected the spatial distribution of silt and clay which decreased with distance from the mouth.

Short-Term Intensive Study:

Sampling at the 178 stations located throughout the lower portion of the Charleston Harbor estuary yielded an updated perspective of sediment distribution within this region (Figure VI.2). Previous studies (Neiheisel and Weaver, 1967; Colquhoun, 1972; Van Niewenhuise *et al.*, 1978) provided the bases for temporal comparisons highlighting rediversion. Topographic, hydrographic and anthropogenic influences, as well as interriverine patterns are evident from our study results.

The harbor basin surficial sediments have been extensively studied due to Charleston's prominence as a seaport and the related shoaling problems which led to rediversion. Charleston Harbor sediments originate from both fluvial and marine sources and their spatial distributions throughout the lower harbor have been described in detail with the use of natural tracers (Neiheisel and Weaver, 1967; Van Niewenhuise *et al.*, 1978). Net effects of rediversion upon bed material have yet to be fully determined; however, results of this sampling effort suggest that spatial distribution of sediment types is very similar to that described in previous studies (Neiheisel and Weaver, 1967; Colquhoun, 1972). Submarine features, such as the Sullivans Island spit (with its littoral sand) and the large southwestern shoal of fluvial silt and clay, were still in existence. The areal coverage of these features had not been altered greatly either. Medium sand was consistently the dominant component of samples retrieved from the north and northeastern basins (Figures VI.8 and VI.9). A graded mixture of sand with silt and clay proceeded west toward the southwestern shoal where a predominance of silt became central to this feature

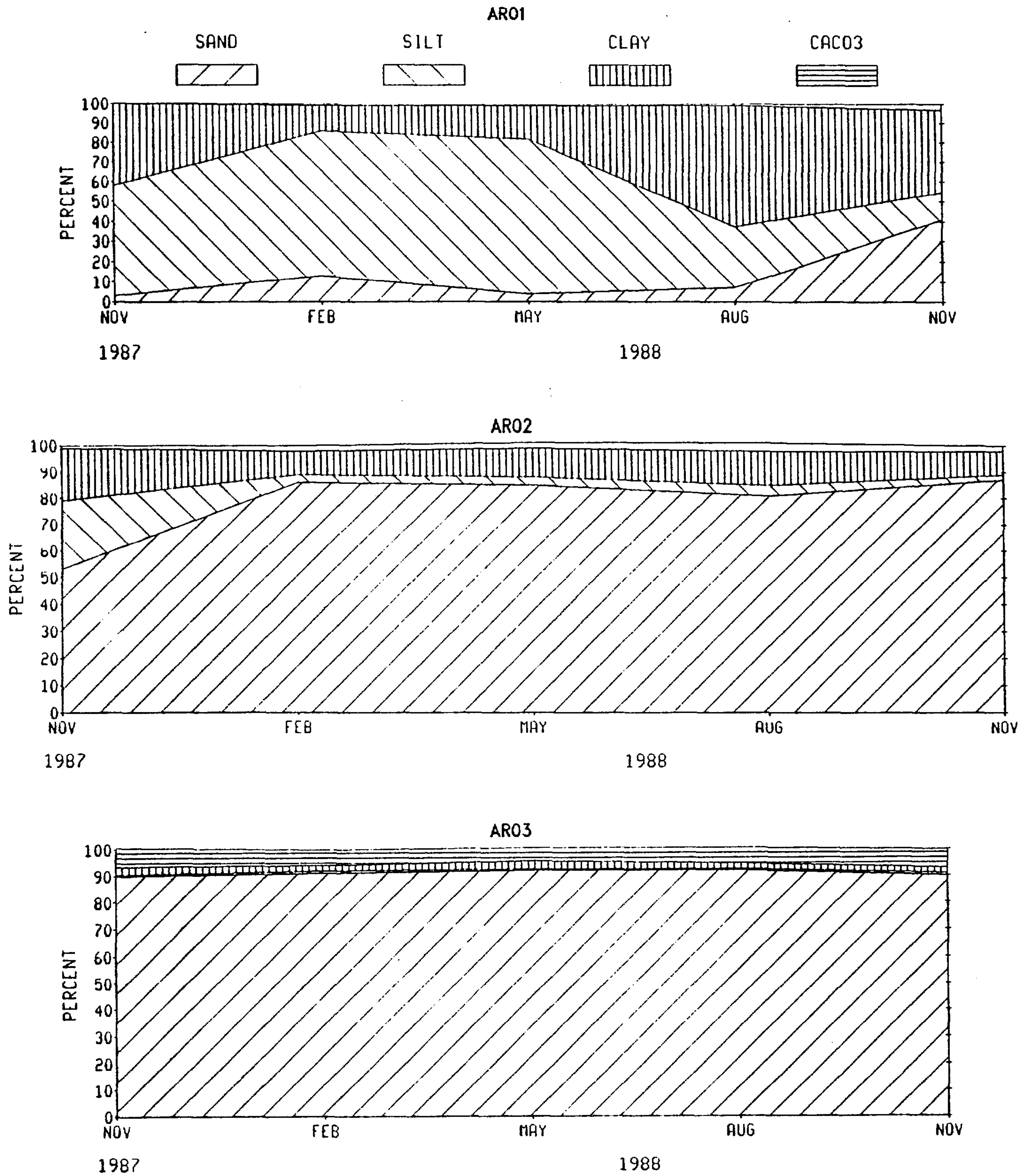

Figure VI.7. Percentages of sand, silt, clay and calcium carbonate at each of the Ashley River sites.

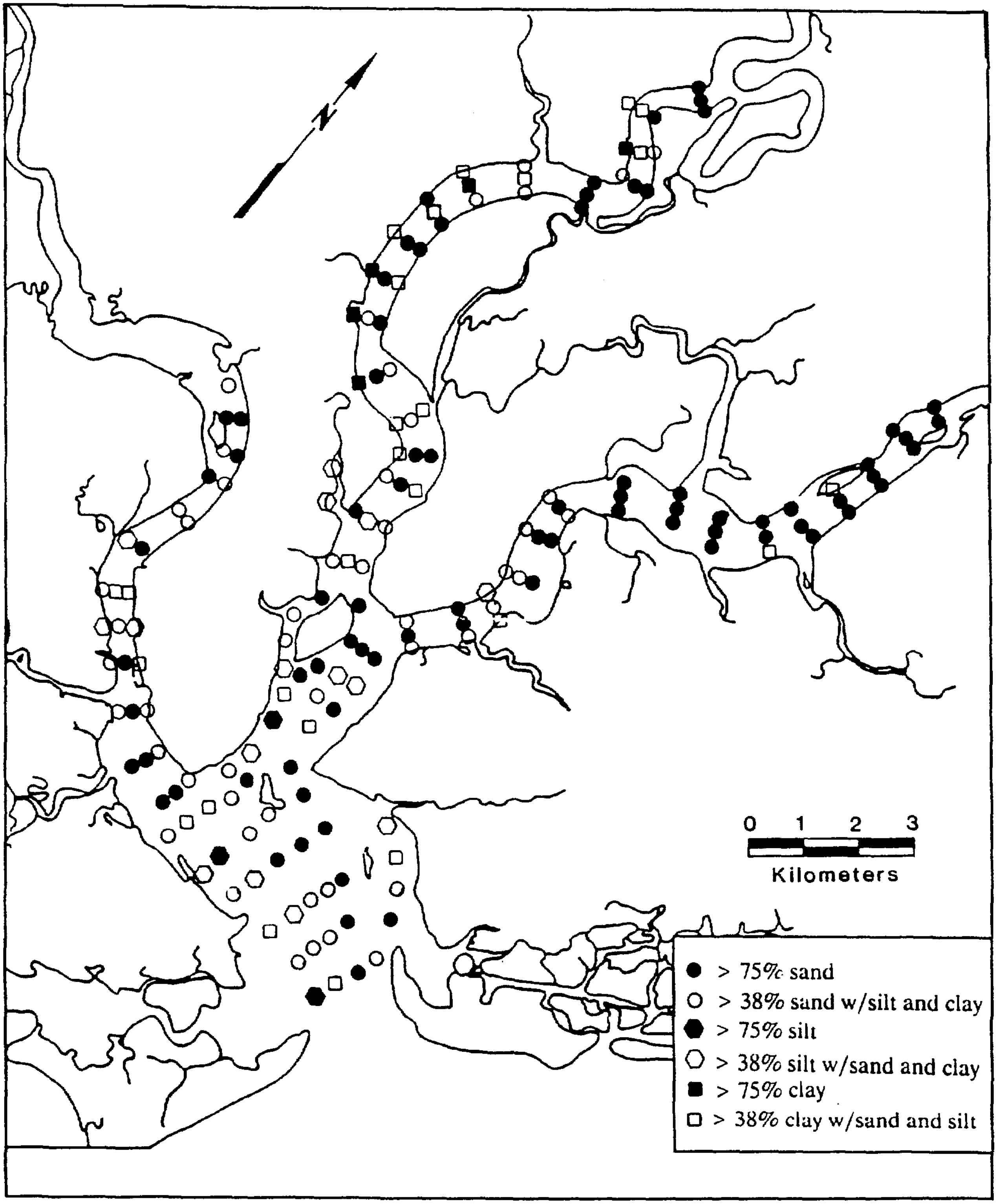

Figure VI.8. Distribution of sediment types in the lower Charleston Harbor estuary during July, 1988.

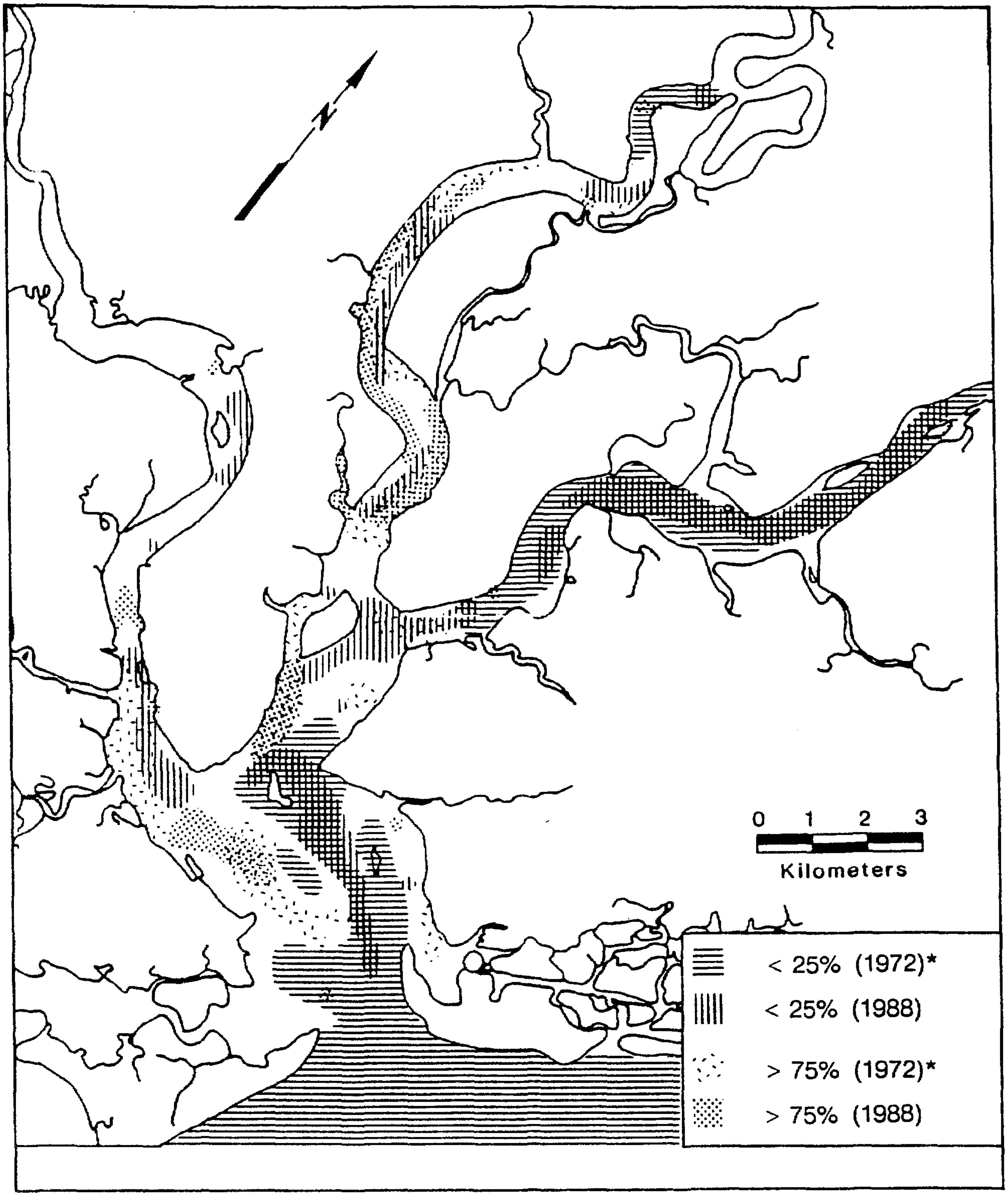

Figure VI.9. Comparison of percent silt and clay in surficial sediments of the lower Charleston Harbor estuary.
*Colquhoun, 1972

with a further gradient towards the finer grained clay material at the mouth of the Ashley River. Hydrodynamic characteristics responsible for deposition in these two areas may have been altered sufficiently by rediversion; however, sediment accumulated prior to rediversion and continued channel deepening may have offset any substantial effects thus far. Conversely, upper harbor basin sediments surrounding Drum Island reflected more apparent shifts in grain size composition. Prior to rediversion, the regions to the west and northeast of the island were characterized by fine silt and clay material. However, these areas proved to be more sandy in nature during our post-rediversion sampling. The area of silt and clay between Drum Island and the State Ports Authority once included all of Town Creek but was restricted to the lower reach in this study. Similarly, while sediments at the confluence of the Wando River and the harbor basin were previously composed of particles measuring less than 63 microns, it now appears to be predominantly fine to medium sand. These sandy sediments extend southward through the navigational channels to the Sullivans Island spit, covering an extensive region along the north side of the harbor basin, Shipyard Creek in the Cooper River and Hobcaw Creek in the Wando. These findings are consistent with Van Nieuwenhuise's (1978) interpretations of the marine sand distribution in the navigational channels and Meade's (1969) conclusion that marine sands are migrating landward. However, the intended disruption of hydrographic stratification may also result in a greater percentage of sand-sized particles by allowing silt and clay to be flushed seaward. In addition, medium sand was concentrated in Rebellion Reach near the Sullivans Island spit, while finer sand dominated the rest of the harbor.

The distribution of organic matter in the Charleston Harbor estuary was mapped by Colquhoun (1972). He noted that organic concentrations greater than 2% were "confined at or seaward of high human activities" prior to rediversion. In July 1988, these concentrations were found in a more extensive region, including all stations at the mouth of the harbor and the Mount Pleasant Channel. Organic content was lower, however, in the Middle Ground south of Shutes Folly Island than it had been in Colquhoun's (1972) study (Figure VI.10). Organic matter concentrations of 5% by weight appeared to be associated with anthropogenic influences such as the port facilities and the Shem Creek docking complex. Similar concentrations were also found in the southwestern shoals of fine-grained bed material, supporting Colquhoun's earlier observations for this region. The apparent areal expansion of organic matter within this system deserves further investigation. Reduced flow since rediversion may result in reduced flushing of man-made and natural toxicants. The tendency of organic matter to absorb heavy metals would compound this problem making the metals more readily available to certain species of benthic organisms as well as the general ecosystem (Forstner and Wittmann, 1983).

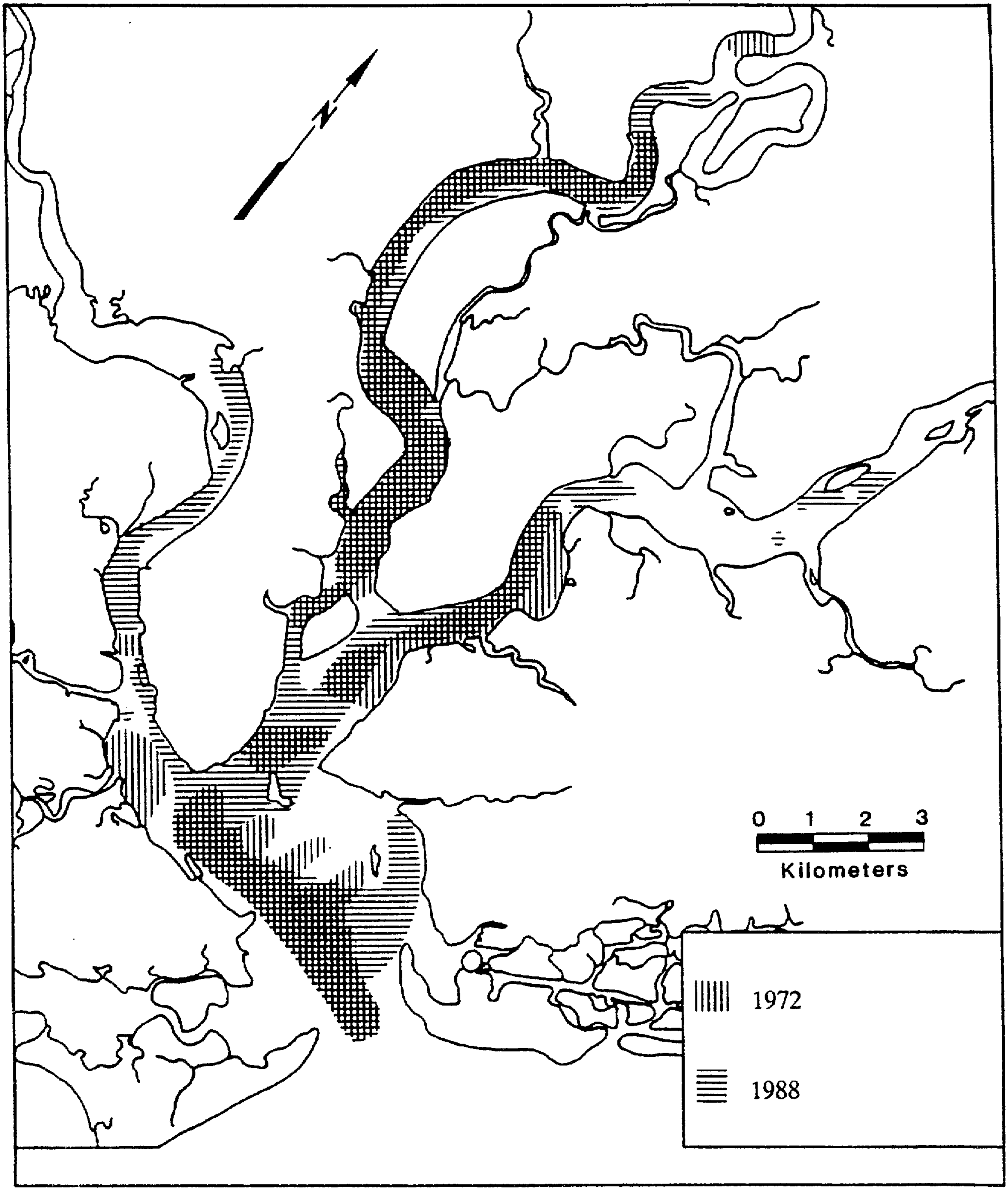

Figure VI.10. Comparison of organic matter distribution (2% by weight) in surficial sediment of the lower Charleston Harbor estuary. Note: The upper Ashley River was not mapped in 1972.
*Colquhoun, 1972

Calcium carbonate accounted for 5% by weight of most samples collected in the harbor (Figure VI.11). The greatest concentrations of calcium carbonate occurred at stations CH55 and CH57 (43.8 and 32.5%, respectively). These two stations are located on opposite sides of Drum Island where there was an abundance of whole and broken shell material. Calcium carbonate was also prevalent in other areas of the estuary which may have been subject to dredging and erosion of the Oligocene Cooper Marl stratum. This stratum is rich in foraminifera and other microfossils dating to the early Oligocene epoch (Colquhoun, 1972). Qualitative assessments of samples retrieved from the lower harbor stations suggest that calcium carbonate encountered throughout this region originated primarily from the presence of shell material rather than deposits of the Cooper Marl.

Bed material of the Cooper River has also been studied extensively as a result of the need to maintain navigable channels. Prior to 1985, the flow rate of the Cooper River was considered to be 98% greater than the Ashley and Wando Rivers combined, and was therefore the only major source of fluvial sediment in Charleston Harbor (Van Nieuwenhuise *et al.*, 1978). Furthermore, as a result of increased freshwater flow following the original diversion, the saltwater wedge and associated sediment trap were more extensive in the Cooper River prior to 1985. Simmons (1966) estimated that the saltwater wedge extended to about the Charleston Navy Yard in the Cooper River prior to rediversion.

The bottom sampling and seismic survey conducted by Colquhoun (1972) provide the most extensive database for comparison with post-rediversion conditions in the Cooper River. The general distribution of surficial sediment texture which he described was roughly equivalent to that observed in July 1988 (Figure VI.9). Regions in which hydrographic energy was obstructed by piers and docking facilities, such as the Navy Base, Naval Shipyard and industrial centers, displayed the greatest concentrations of fine-grained material. With the exception of CR39, which was near the mouth of Filbin Creek, all stations along the industrialized west bank of the Cooper River were dominated by clay (Figure VI.8). While unconsolidated material was characteristic of the Naval complex, densely compacted clay, indicative of a more scoured bed, was encountered within the North Charleston Port Terminal basin. A loose mixture of silt and clay was collected at stations in Shipyard Creek, the only tributary to the Cooper River encompassed in the short-term study sampling array. The fine material in that creek accounted for 96-99% by weight of the samples collected. Sediments were sandier in other portions of the Cooper River than were noted in previous studies. The distribution of material less than 63 microns is roughly equivalent to that presented by Colquhoun (1972). However, sand dominated sediments from along the east bank of the Cooper River across from the Naval Shipyard to the region opposite

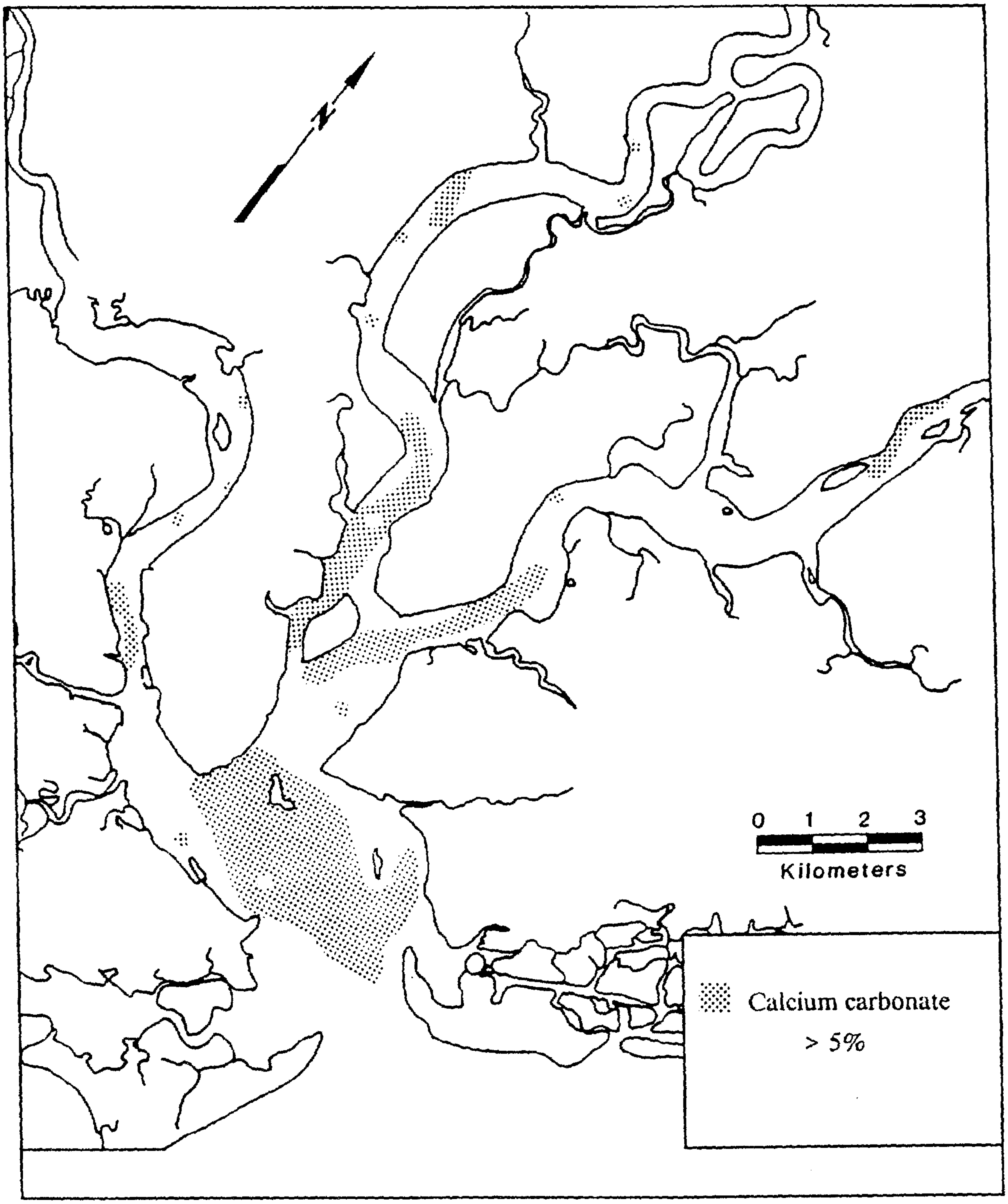

Figure VI.11. Calcium carbonate distribution in surficial sediments of the lower Charleston Harbor estuary during July, 1988.

Westvaco. Sediments collected from this area during the 1988 sampling period were composed primarily of fine to coarse-grained sand while a more even mixture of sand, silt and clay was detected by Colquhoun. Station CR02, which was located in this region and sampled over the four-year period displayed no variation in sediment type relative to rediversion or to season. It is possible that the trend towards a sandier bottom type predates rediversion and results from other changes in the system. Sand grain-size distribution indicated that mostly well-sorted, fine to medium sand continued from the harbor to Filbin Creek. Above this point, sorting and grain size of the sand fraction were erratic (Figure VI.12).

The distribution of organic matter within the Cooper River had also expanded in comparison to that described by Colquhoun (1972). We observed organic matter (>2%) throughout the entire study area (Figure VI.10); however, concentrations >10% were closely associated with the occurrence of clay.

As in the harbor basin, calcium carbonate distribution within the Cooper River displayed no direct relationship to live communities of oysters or the clam, *Mulinia lateralis*; however, remnants of fossil mollusks are a part of sediments found in this region (Figure VI.11, also see Chapter IX). Calcium carbonate concentrations were not distributed according to sand size or sorting coefficients in the Cooper River, either. The Cooper Marl lies beneath a thin stratum of Holocene sands in many locations of this river. Thus, it is likely that scouring and dredging of this fossiliferous deposit has contributed significantly to the calcium carbonate concentration noted in this river, also.

Surficial sediments of the Wando River displayed the greatest spatial uniformity compared with other portions of the estuary (Figure VI.8). Relic sand deposits have characterized this region in previous investigations (Colquhoun, 1972; Van Nieuwenhuise, 1978). In the current study, sandy sediments were common at the mouth of this river, as well. While silt and clay once dominated this area, we noted a varied mixture of fine to coarse sand with lower percentages of silt and clay during 1988. Sediments at station WR01, sampled in the long-term study, were somewhat sandier following rediversion; however, even prior to that event, sand was the dominant component at that site. The only predominantly silty station, located in the Wando River, (WR16) was situated near a dredge spoil discharge pipe for the Daniel Island disposal site. The clayey silt observed at this site may represent sediments which were resuspended in the dredge spoil "dewatering" process. Clay was found to be dominant at only three isolated stations in the Wando River. Above the Wando Port Terminal, surficial sediments were predominantly well-sorted medium sand with little variation in sorting and grain size (Figure VI.12).

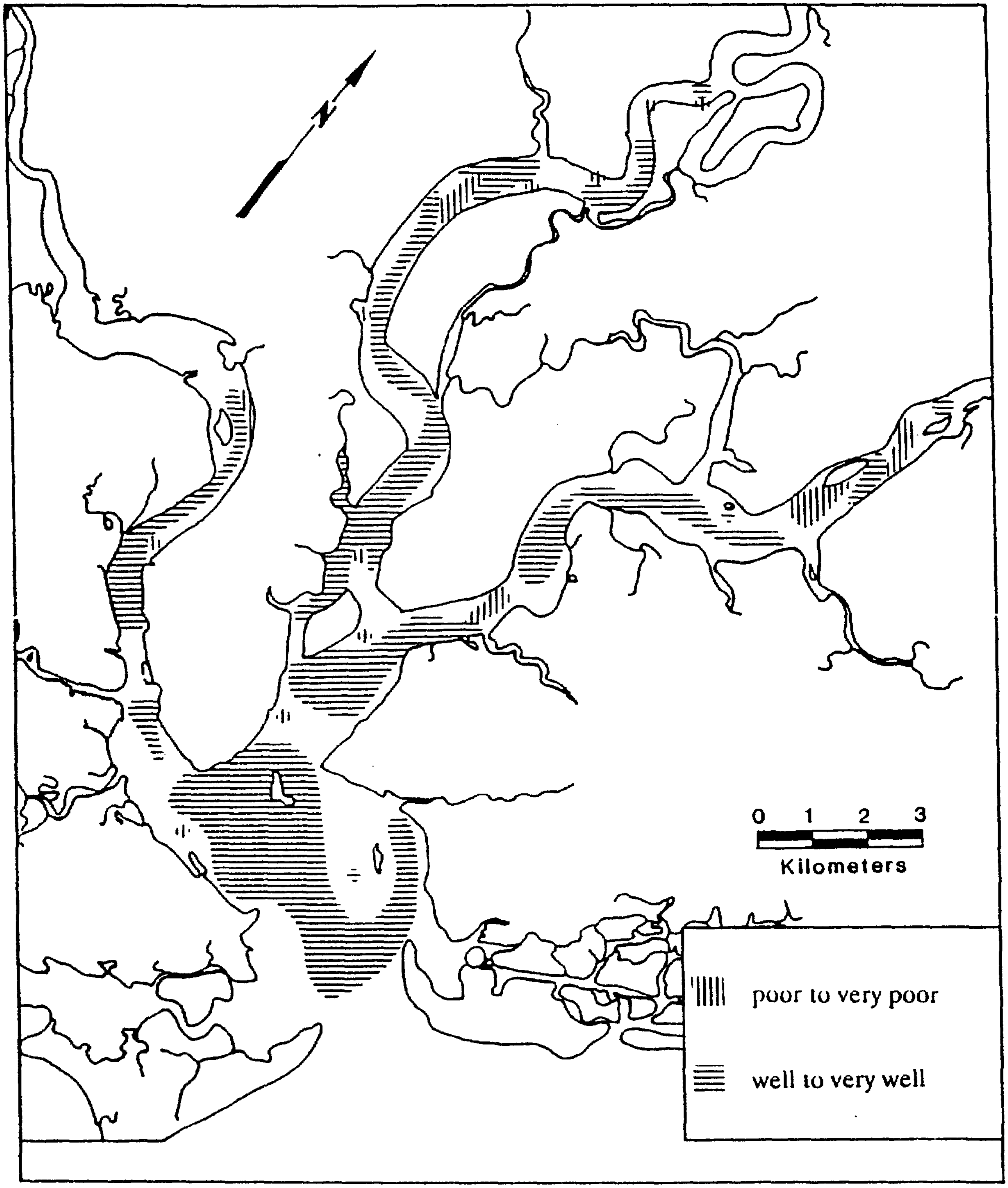

Figure VI.12. Sand sorting distribution in sediment of the lower Charleston Harbor estuary during July, 1988.

Organic content of the surficial sediments in the upper Wando River was greater than described by Colquhoun in 1972 (Figure VI.10). He found that organic matter was concentrated at or seaward of the Wando Port Terminal, whereas we observed sporadic concentrations up to Juba Island. These locations were also associated with the higher occurrences of silt and clay.

Sources of calcium carbonate abound within the Wando River. Extensive intertidal oyster banks are present along Beresford Creek and the eastern shore of the river. A subtidal concentration of oysters also exists from Juba Island upriver to Detyens Shipyard. In addition, microfossils associated with the Cooper Marl contributed substantially to the amount of calcium carbonate at one station. The sample from station WR20, a channel site opposite the Wando port facilities, was found to contain 10% calcium carbonate derived from Oligocene foraminifera, with no shell hash evident. Shell hash was the major source of calcium carbonate at other stations which occurred in two separate regions (Figure VI.11). One area extended from the port facility to the mouth and was associated with the medium to coarse sand component. The other was located throughout the subtidal oyster beds of the upper Wando River. Bottom sampling with the Ponar grab was hindered considerably by oyster shells and clusters which prevented full closure of the mechanism and, consequently, loss of most of the sample. Sampling was repeated until an intact bottom sample was retrieved; however, this usually precluded the capture of large shell material. Therefore, it is likely that percentages of calcium carbonate for this area, as a whole, were considerably greater than our data suggest (Appendix VI.2).

Comparable data on the bottom sediments of the Ashley River are primarily limited to Colquhoun's survey in 1972. However, his bottom sampling was limited to the lower Ashley, seaward of the Highway 17 bridge. Formerly a region in which silt and clay accounted for more than 75% of the sample weight, the channel and areas along the Charleston peninsula were characterized, respectively, by sand and sandy silt in 1988 (Figure VI.9). The expanded occurrence of sandy sediment throughout the lower Charleston Harbor estuary may be due to pre-rediversion conditions in which hydrographic stratification dictated landward transport of bed material. Net upriver transport of marine sands (Meade, 1969; Van Nieuwenhuise *et al.*, 1978) probably continued for the thirteen years following the last intensive assessment of surficial sediments in the Charleston Harbor estuary (Colquhoun, 1972).

The surficial sediments upriver from the Highway 17 bridge to approximately Orange Grove Creek were composed primarily of silt and clay mixed with well-sorted, fine-grained sand. This graded into a sandy substrate which extended to the limit of our study area. Sand throughout this region ranged from poor to very well sorted and was coarser than sand observed lower in the river (Figure VI.12). The seismic profile conducted by Colquhoun (1972) showed a similar mixture of "sands, clays and sludge" in the channel above the Highway 17 bridge.

Organic matter was widely distributed throughout the Ashley River (Figure VI.10). Concentrations were primarily associated with the occurrence of silt and clay as noted for the other river systems. Though organic content percentages of Ashley River sediments were greater than in the Wando River, they were substantially less prevalent than in the naval/industrial corridor of the Cooper River.

The Cooper Marl contributes to Ashley River bed material as well. Station AR13 contained the greatest concentration of calcium carbonate encountered in the Ashley River (24.8%). This station had an abundance of microscopic foraminifera. Most of the other Ashley River stations with concentrations of calcium carbonate in excess of 5% were associated with fine-grained material (Figure VI.11). Oyster shell found along the banks of the Ashley River to approximately 200 meters upriver from Orange Grove Creek, may have contributed to the calcium carbonate content of sediments in this study. *Mulinia lateralis* occurred in high numbers at several stations; however, calcium carbonate percentages were not elevated at those sites.

SUMMARY

1. Surficial sediments of the Charleston Harbor estuarine system were less variable proceeding upriver in terms of the textural and mineralogical parameters analyzed. Wide, non-seasonal fluctuations related to spatial variability in sediment type were characteristic of the lower harbor stations. Sediments were sandier proceeding upriver, as well. However, mean grain size of the sand fraction was not closely related to river mile.

2. An average increase in the percentage of silt and clay was found at all four Cooper River stations during the three-year post-rediversion period, particularly at station CR01.

3. The short-term intensive benthic survey of the Charleston Harbor estuary revealed a general increase in the occurrence of predominantly sandy sediments in comparison to previous studies. The Sullivans Island spit, a distinct topographic feature of the harbor basin historically, appeared to have directed sandy sediment upestuary along the channel to include a large region adjacent to Drum Island. This trend toward

sandier sediments was evident at the mouths of the Ashley and Wando Rivers, as well as along the western bank of the lower Cooper River.

4. Optimum conditions for sorting of the sand fraction were found throughout the harbor basin and lower reaches of the three rivers. The degree of tidal influence did appear to have a recognizable impact upon sorting coefficients, however, as exhibited by more poorly sorted sediments in the upper reaches of the estuary.

5. Areas of predominantly fine material such as the large shoal to the southwest of Shutes Folly Island displayed less pronounced spatial or temporal change. A consistent association between organic matter and fine material was detected at these locations. Organic matter was more prevalent throughout the Charleston Harbor estuary in comparison to historical data. Though the greatest concentrations were found near areas of low hydrodynamic energy, sediments throughout most of the study area contained at least two percent organic matter by weight. The greatest percentages occurred in the Cooper River and the fluvially created western shoal of the harbor basin.

7. A comparison of sediment distribution derived from the long-term seasonal and short-term intensive studies highlights the spatial variability evident throughout the Charleston Harbor estuary. The long-term study stations were indicative of a broader areal distribution in the more homogeneous regions only (WR02 and WR03). Though the short-term study stations were often within 150 m of one another, substantial differences in sediment type were observed between adjacent sites. Temporal variability attributed to minor discrepancies in site relocation further reflects the patchy nature of surficial sediments within the Charleston Harbor estuary. The interaction of numerous physical and biological variables have resulted in a complex distribution of surficial sediments within this system.

CHAPTER VII

BENTHIC MACROFAUNAL COMMUNITIES

by

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INTRODUCTION

Prior to 1984, surveys of benthic macrofaunal communities in the Charleston Harbor estuary were very limited. One of the earliest known studies was conducted in 1965 by the Federal Water Pollution Control Administration (1966). Although the primary emphasis of that study was to evaluate water quality in the estuary, benthic samples were collected at 20 sites during one sampling period. The samples indicated that the benthic communities were adversely affected by pollution in several reaches of the harbor system. From 1973 to 1974, Calder and Boothe (1977) collected samples from eight sites in the harbor system as part of a larger survey of estuarine benthic assemblages throughout the state. Published data on the macroinfauna from these stations is limited to one or a few seasons at four sites located in the Cooper River, two sites located in the harbor basin, and one site each located in the Wando and Ashley Rivers. Additional surveys conducted in the estuary since then include very limited assessments of the benthos in the middle and upper reaches of the Cooper River in 1975 and 1982 (Dames and Moore, 1975; Enwright Associates, 1977; Williams, 1984) and a quarterly assessment at eight sites in the lower Wando River from 1981 to 1984 (Enwright Laboratories, Inc., 1984).

Due to the paucity of data available on benthic communities present in the Charleston Harbor estuary, two studies were initiated in 1984 and 1988 to obtain additional data on the temporal and spatial variability of these trophically important taxa. Specific objectives of the study begun in 1984 were to:

1. describe seasonal and site-related patterns of benthic community structure at ten stations located in the harbor basin, Cooper River and Wando River over a four-year period,

2. evaluate changes in the abundance and distribution of benthic macrofauna following the Cooper River Rediversion Project, and

3. describe seasonal and site-related patterns of benthic community structure at three sites in the Ashley River over a one-year period following rediversion.

Specific objectives of the study begun in 1988 were to:

1. characterize the spatial distribution of benthic fauna at 178 sites located throughout the lower portion of the estuary during one season, and

2. relate distribution patterns of the benthos to various natural and anthropogenic environmental factors.

LONG-TERM SEASONAL STUDY

METHODS

Benthic macrofaunal communities were sampled seasonally over a four-year period at 10 sites located throughout the Cooper River, Wando River and Charleston Harbor (Figure VII.1). This portion of the study included one year of sampling prior to rediversion (November 1984 - August 1985) and three years of sampling after rediversion (November 1985 - November 1988). In addition, three sites in the Ashley River were sampled seasonally over a one-year period (November 1987 - November 1988) to provide a more comprehensive overview of the benthic macrofauna throughout the Charleston Harbor estuary.

Three replicate 0.05-m^2 Ponar grab samples were collected during each site visit, along with measurements of surface and bottom water temperature, salinity and dissolved oxygen. After removing a small subsample of sediment from one of the three replicate grabs for subsequent grain size and mineralogical analyses, each sample was washed through a 0.5-mm mesh sieve and the organisms retained on the sieve were preserved in a solution of 10% formalin containing the vital stain rose bengal. In the laboratory, these organisms were sorted and identified to the lowest practicable taxonomic level.

Species diversity was measured for each set of pooled replicate samples using the Shannon-Weaver (H') index (Pielou, 1975). The two components of this diversity index,

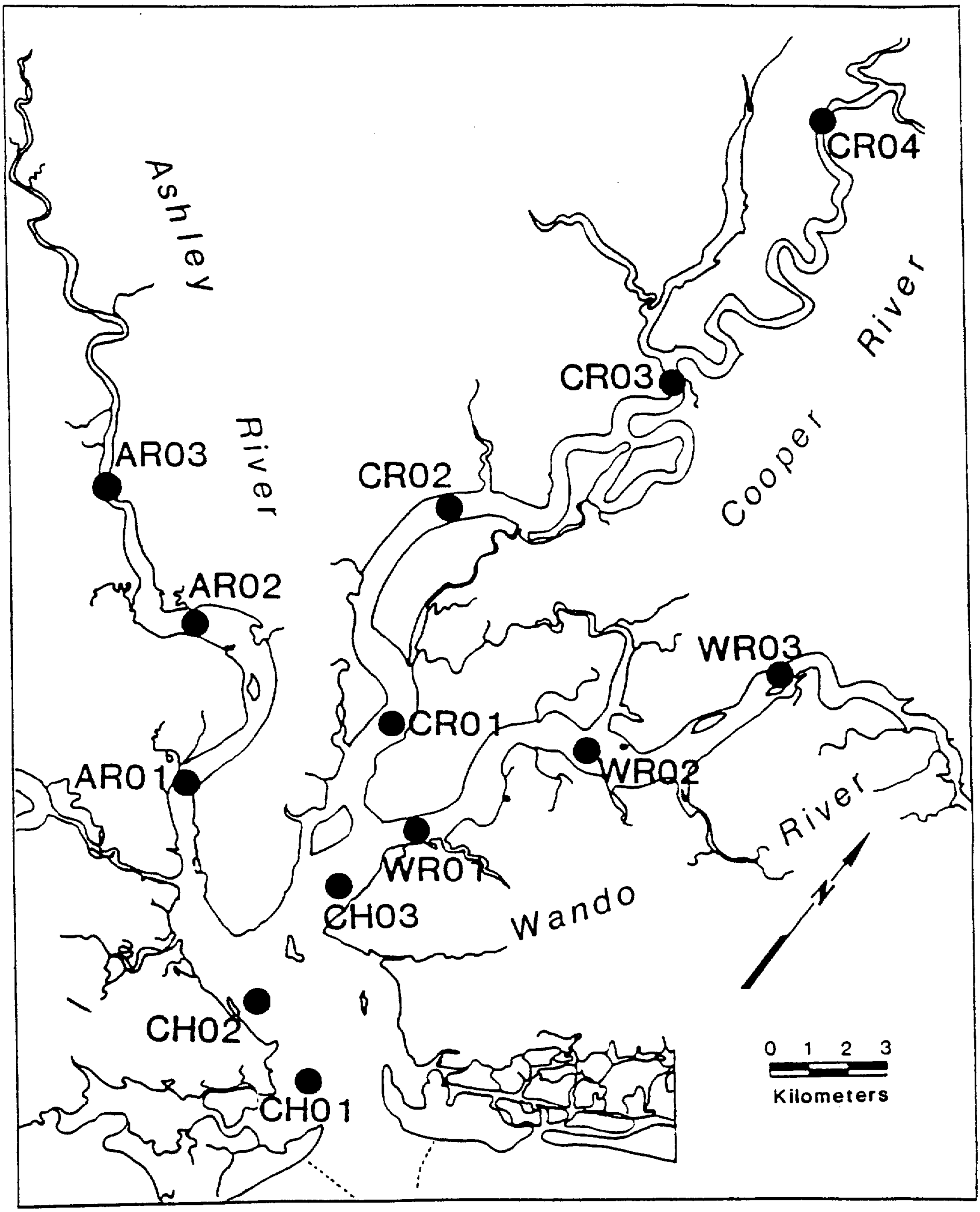


Figure VII.1. Sampling locations for the four-year study of benthic macrofauna in the Charleston Harbor estuary.

species richness (S-1/ln N) and species evenness (H'/H' max), were also calculated separately (Margalef, 1958; Pielou, 1975). Normal cluster analyses were used to group pooled replicate samples within each river system and the harbor basin on the basis of similarities in their species composition as defined by the Bray-Curtis (1957) similarity coefficient. Inverse cluster analyses were used to group species on the basis of their distributions among all sets of pooled replicate samples within a particular river or the harbor. Only those species which accounted for at least 0.1% of the total faunal abundance in a given river or in the harbor basin, and which occurred in at least three collections throughout the four-year sampling period, were included in the normal and inverse cluster analyses. Species abundances were first log-transformed [$\log_{10}(x+1)$, where x = number of individuals] to lessen the influence of extremely abundant species in these analyses. The clustering algorithm chosen was the flexible sorting strategy of Lance and Williams (1967), with the cluster-intensity coefficient (β) set at -0.25. Nodal analyses (Boesch, 1977a) were used to interpret normal and inverse cluster analyses by expressing the degree of species/site group coincidences in terms of the classic community concepts of constancy and fidelity. Nodal constancy is a measure of how consistently the members of a particular species group occur among the stations of a particular site group, while nodal fidelity is a measure of the degree to which a particular species group is restricted to a particular site group. Formulae for these expressions are given by Boesch (1977a).

Three-way (Model I) analyses of variance were used to compare $\log_{10}$-transformed abundances and total numbers of species among pooled replicate samples representing each year, season and site of collection within the Cooper River, Wando River and Charleston Harbor. Two-way (Model I) analyses of variance were used to compare the same two variables among seasons and sites within the Ashley River. The Student-Newman-Keuls multiple range test was used in *a posteriori* comparisons of group means for each of the main effects.

RESULTS AND DISCUSSION

Cooper River:

Over the course of four years, 8,028 macrofaunal organisms representing 122 taxa were collected at the 4 stations located throughout the Cooper River (Appendix VII.A). Mean abundances ranged from a low of 3.67 individuals/grab at station CR02 in Summer 1988 to a high of 286.33 individuals/grab at station CR01 in Spring, 1988. Despite the wide range of abundances, this variable exhibited no apparent trends with respect to site, season, or year of collection (Figure VII.2). Similarly, there was no obvious difference in the total abundance of benthic organisms before versus after rediversion. These observations are

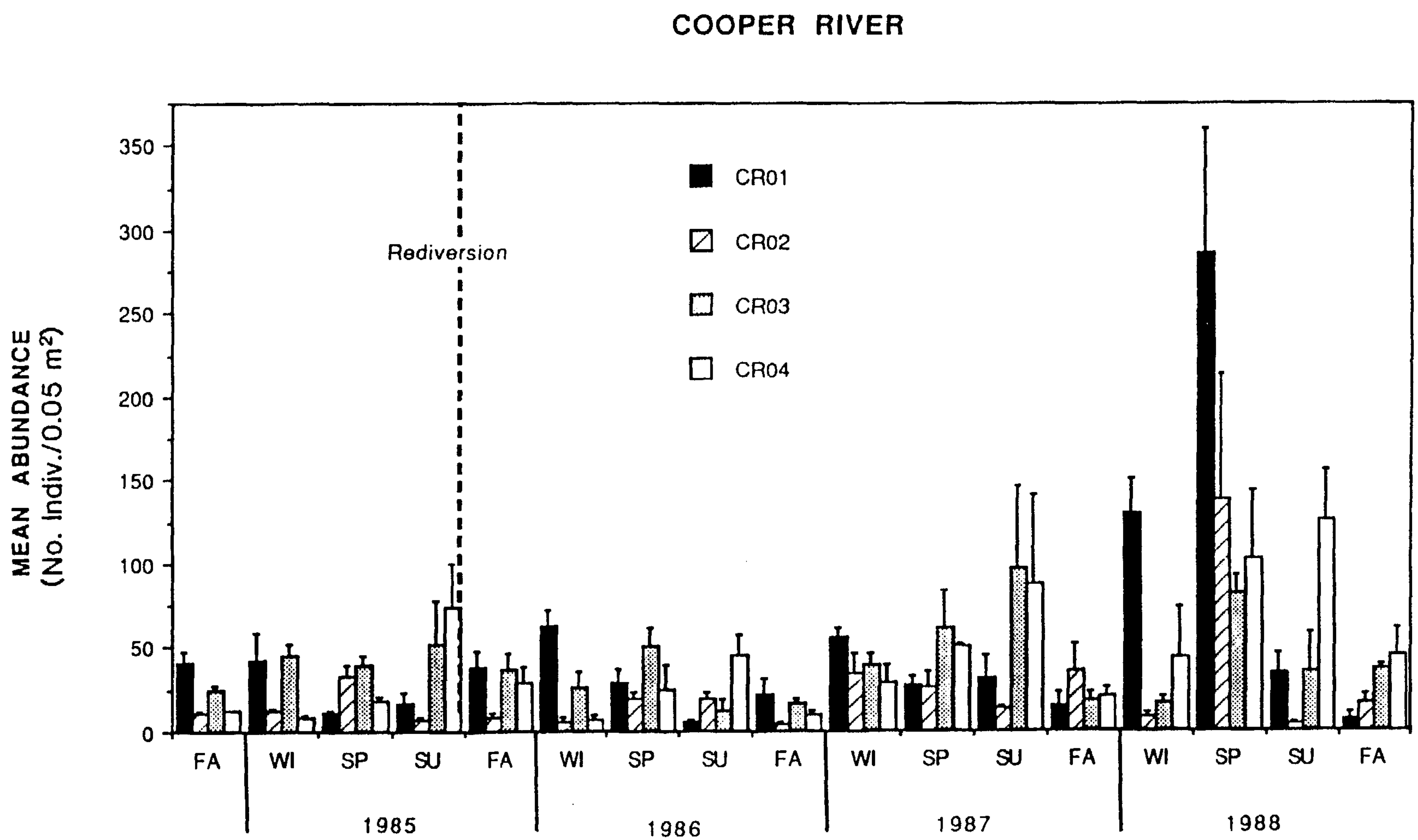


Figure VII.2. Mean abundance of macrofaunal organisms collected at each of the Cooper River sampling sites during each season of the four-year study. (Vertical lines represent one standard error of the mean).

supported by the results of a three-way analysis of variance, in which the three sources of variation tested (site, season and year) displayed highly significant first- and second-order interaction effects (Table VII.1). As a result of these interactions, the *a posteriori* comparisons of main effect group means (Table VII.2) are rather difficult to interpret and require some qualification. For example, although the Student-Newman-Keuls multiple range test showed mean abundances to be significantly higher in spring, it is apparent from Figure VII.2 that this is true only for particular sites during particular years (most notably, the two downriver sites CR01 and CR02 in 1988). Similarly, although CR02 had significantly lower abundances than the other three sites overall, it had higher abundances than one or more sites during seven of the 17 sampling periods.

Species diversity (H') values also ranged widely, from a low of 0.55 bits/individual at station CR02 in Winter, 1987 to a high of 3.47 bits/individual at CR01 in Spring, 1986 (Figure VII.3a). In general, however, these values showed no consistent trends with respect to site, season or year of collection. The same inconsistency was true of species evenness (J') values as well (Figure VII.3b). Species richness, however, was generally higher at the two stations closest to the mouth of the river (CR01 and CR02) than it was at the two stations further upriver (Figure VII.3c). This tendency toward greater species richness near

Table VII.1 Results of three separate three-way (Model I) analyses of variance comparing mean numbers of macrofaunal organisms per grab sample among sites, seasons and years within the Cooper River, Wando River and Charleston Harbor.

Dependent variable: Log_{10} (total no. indiv. + 1)

Source of Variation	Cooper River df	Cooper River F_s	Wando River df	Wando River F_s	Charleston Harbor+ df	Charleston Harbor+ F_s
Year	3	8.09 ***	3	4.19 **	3	12.28 ***
Season	3	11.15 ***	3	6.61 ***	3	7.39 ***
Site	3	14.07 ***	2	15.35 ***	1	99.11 ***
Year x Season	9	5.93 ***	9	5.10 ***	9	6.42 ***
Year x Site	9	1.93 N.S.	6	1.70 N.S.	3	15.86 ***
Season x Site	9	8.37 ***	6	2.00 N.S.	3	3.58 **
Year x Season x Site	27	2.42 ***	18	2.49 **	9	3.70 ***

\+ CH03 not included
* $p \leq 0.05$
** $p \leq 0.01$
*** $p \leq 0.001$

Table VII.2. Results of *a posteriori* comparisons (Student-Newman-Keuls Tests) of main effect group means+ for the Cooper River, Wando River and Charleston Harbor. (Dependent variable: Log_{10} [total no. indiv. +1]).

	Dependent variable: Log_{10} (total no. indiv. +1)											
Source of Variation	Cooper River				Wando River				Charleston Harbor++			
Year[1.]	17.65 μ_{1986}	20.07 μ_{1985}	25.70 μ_{1987}	33.97 μ_{1988}	17.37 μ_{1987}	17.51 μ_{1988}	20.39 μ_{1985}	30.19 μ_{1986}	44.03 μ_{1985}	60.14 μ_{1986}	75.57 μ_{1987}	123.95 μ_{1988}
Season[2.]	16.74 μ_{FA}	21.87 μ_{SU}	22.41 μ_{WI}	37.63 μ_{SP}	14.41 μ_{FA}	18.58 μ_{WI}	22.22 μ_{SU}	31.36 μ_{SP}	42.51 μ_{FA}	81.36 μ_{WI}	82.84 μ_{SU}	86.68 μ_{SP}
Site[3.]	13.43 μ_{CR02}	25.04 μ_{CR04}	29.92 μ_{CR03}	30.56 μ_{CR01}	13.20 μ_{WR02}	21.71 μ_{WR03}	31.23 μ_{WR01}		37.90 μ_{CH01}	130.94 μ_{CH02}		

\+ Group means are re-transformed to the linear scale; those means connected by underlines are not significantly different at p=0.05.

++ CH03 not included

1. Cooper River: n=48; Wando River: n=36; Charleston Harbor: n=24
2. Cooper River: n=48; Wando River: n=36; Charleston Harbor: n=24
3. Cooper River: n=48; Wando River: n=48; Charleston Harbor: n=48

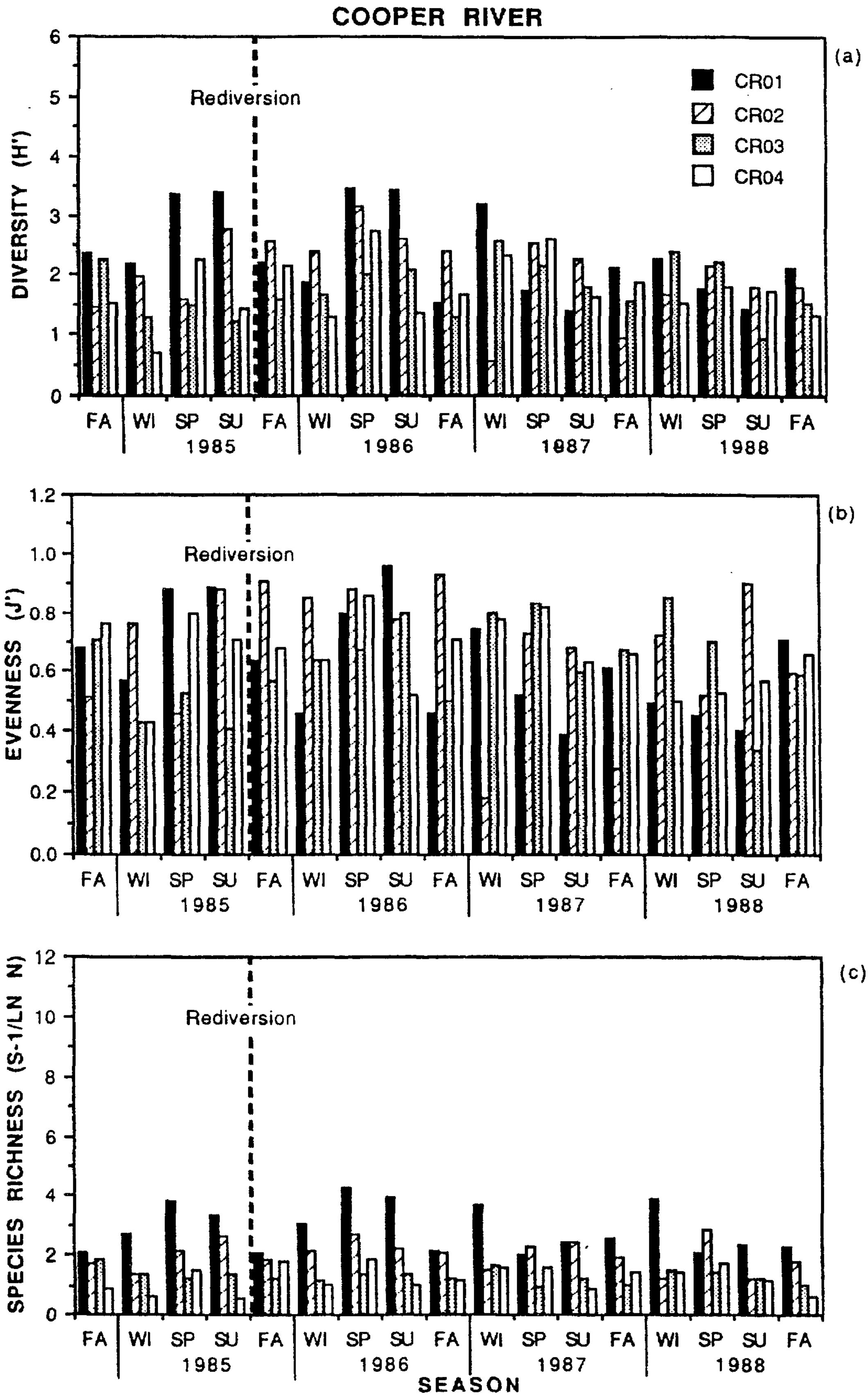


Figure VII.3. Species diversity (H'), evenness (J') and richness (S-1/ln N) values for benthic macrofaunal samples collected at each of the Cooper River stations during each season of the four-year study.

the mouth of the river is confirmed by the results of a second three-way analysis of variance performed on the total number of unique species in each set of pooled replicate grab samples (Table VII.3). The variation in species number among sites was clearly greater than it was among years or seasons of collection, although differences among seasons were also highly significant. Once again, however, statistically significant interaction effects complicate the interpretation of *a posteriori* comparisons among main effect group means (Table VII.4). Thus, although species number was significantly greater at CR01 than at CR02 or either of the upriver sites, this was not invariably the case. Similarly, species number, like total abundance, was significantly greater in spring overall, but not at all sites or even in all years.

It is evident from these analyses that neither the total abundance of macrofaunal organisms nor the total number of species in the Cooper River differed significantly between pre-rediversion and post-rediversion sampling periods. The results of cluster and nodal analyses lend further support to the contention that rediversion did not affect the overall community structure in any easily discernible way (Figure VII.4 and V.5; Table VII.5). Furthermore, site-related differences appear, once again, to have been more important than either the year or season of collection in determining the species composition and abundance of organisms.

Table VII.3 Results of three separate three-way (Model I) analyses of variance (without replication) comparing the total number of unique species in each set of replicate samples among sites, seasons and years within the Cooper River, Wando River and Charleston Harbor.

Dependent variable: Log_{10} (total no. species + 1)

Source of Variation	Cooper River df	Cooper River F_s	Wando River df	Wando River F_s	Charleston Harbor+ df	Charleston Harbor+ F_s
Year	3	1.25 N.S.	3	3.22 *	3	1.44 N.S.
Season	3	6.27 **	3	4.43 *	3	5.56 *
Site	3	35.11 ***	2	58.18 ***	1	13.65 **
Year x Season	9	2.27 *	9	2.17 N.S.	9	2.41 N.S.
Year x Site	9	2.30 *	6	1.44 N.S.	3	1.52 N.S.
Season x Site	9	3.11 **	6	2.21 N.S.	3	0.30 N.S.
Year x Season x Site	27		18		9	

+ CH03 not included
* $p \leq 0.05$
** $p \leq 0.01$
*** $p \leq 0.001$

Table VII.4. Results of *a posteriori* comparisons (Student-Newman-Keuls Tests) of main effect group means+ for the Cooper River, Wando River and Charleston Harbor. (Dependent variable: Log_{10} [total no. species +1]).

Dependent variable: Log_{10} (total no. species +1)

Source of Variation	Cooper River				Wando River				Charleston Harbor++			
Year[1.]	8.10 μ_{1985}	8.86 μ_{1987}	9.19 μ_{1986}	9.30 μ_{1988}	11.89 μ_{1988}	14.14 μ_{1987}	15.22 μ_{1985}	16.78 μ_{1986}	19.89 μ_{1985}	22.44 μ_{1988}	24.12 μ_{1987}	25.92 μ_{1986}
Season[2.]	7.91 μ_{FA}	8.33 μ_{SU}	8.55 μ_{WI}	10.75 μ_{SP}	12.48 μ_{FA}	12.48 μ_{WI}	15.22 μ_{SU}	18.05 μ_{SP}	17.20 μ_{FA}	22.44 μ_{SP}	24.12 μ_{WI}	29.20 μ_{SU}
Site[3.]	6.76 μ_{CR04}	7.32 μ_{CR03}	8.77 μ_{CR02}	13.79 μ_{CR01}	9.72 μ_{WR03}	11.30 μ_{WR02}	26.54 μ_{WR01}		19.42 μ_{CH01}	27.18 μ_{CH02}		

\+ Group means are re-transformed to the linear scale; those means connected by underlines are not significantly different at p=0.05.
++ CH03 not included
1. Cooper River: n=16; Wando River: n=12; Charleston Harbor: n=8
2. Cooper River: n=16; Wando River: n=12; Charleston Harbor: n=8
3. Cooper River: n=16; Wando River: n=16; Charleston Harbor: n=16

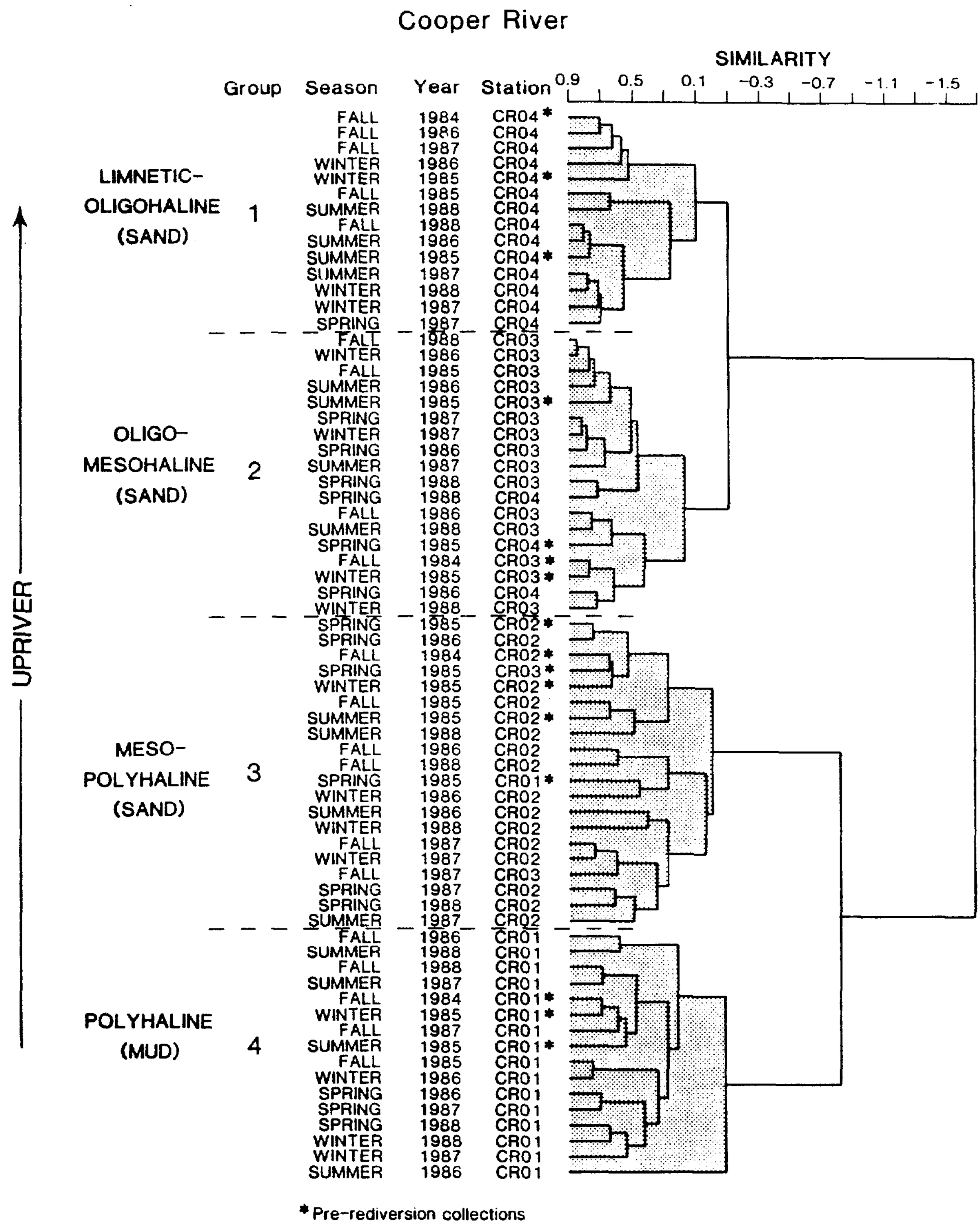


Figure VII.4. Hierarchical classification of Cooper River grab samples generated by a normal cluster analysis.

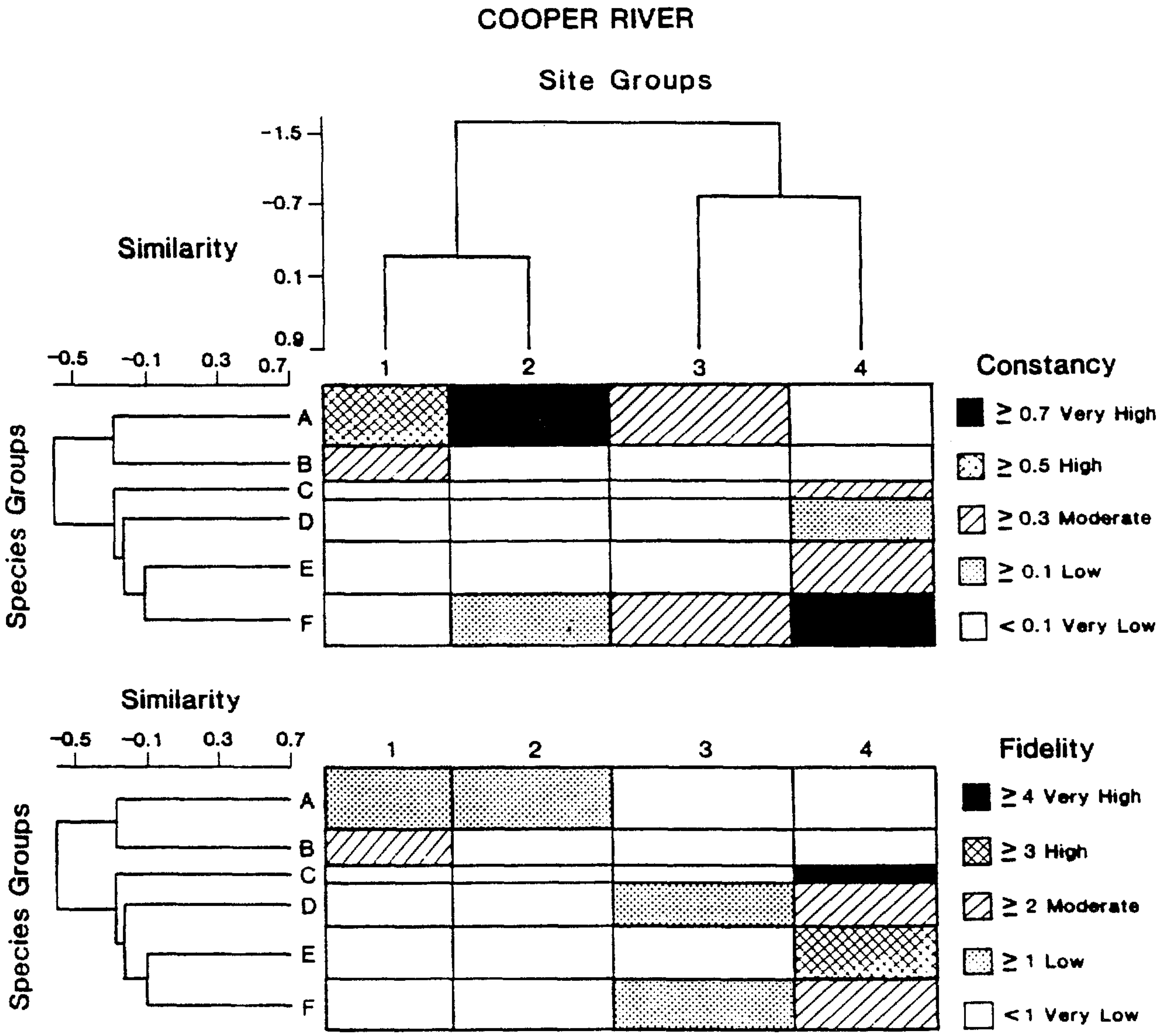


Figure VII.5. Nodal constancy and fidelity diagrams illustrating coincidences between species groups and site groups generated by inverse and normal cluster analyses of Cooper River grab data.

A normal cluster analysis of pooled replicate grab samples generated four discrete site groups, each consisting almost exclusively of collections from one of the four Cooper River sites (Figure VII.4). Samples taken at the two upriver sites (CR04 and CR03) forming groups 1 and 2, respectively, clustered separately from those taken at the two downriver sites (CR02 and CR01) comprising groups 3 and 4. There was no consistent grouping of samples on the basis of year or season of collection, and pre-rediversion

Table VII.5. Species groups generated by an inverse cluster analysis of Cooper River grab samples. (Am = amphipod; B = bivalve; Cu = cumacean; G = gastropod; Is = isopod; P = polychaete).

GROUP A

Lepidactylus dytiscus (Am)
Scolecolepides viridis (P)
Chiridotea almyra (Is)
Gammarus tigrinus (Am)
Monoculodes sp. A (Am)
Chiridotea stenops (Is)
Nematoda

GROUP B

Cyathura polita (Is)
Chironomidae (In)
Oligochaeta
Ceratopogonidae (In)

GROUP C

Glycinde solitaria (P)
Nassarius vibex (G)

GROUP D

Melita nitida (Am)
Corophium lacustre (Am)
Eteone heteropoda (P)
Petricola pholadiformis (B)
Mancocuma sp. (Cu)

GROUP E

Ostracoda B
Leucon americanus (Cu)
Cyclaspis varians (Cu)
Ilyanassa obsoleta (G)
Oxyurostylis smithi (Cu)
Edotea montosa (Is)

GROUP F

Nemertinea
Heteromastus filiformis (P)
Mulinia lateralis (B)
Paraprionospio pinnata (P)
Nereis succinea (P)
Streblospio benedicti (P)

samples did not cluster separately from post-rediversion samples collected from the same site.

Samples from station CR02 were generally more similar to those from CR01 than to those from CR03 or CR04, despite a substantial difference in sediment type between the two downriver sites. Station CR02, like the two upriver sites, was characterized by predominantly sandy sediments; whereas, sediments at station CR01, while more variable in composition, generally consisted mostly of silts and clays (Figure VII.6). Nevertheless, the meso- to polyhaline salinities at station CR02 were much more similar to those at station CR01 than they were to those at the two upriver sites, which were more often in the oligo- to mesohaline range (Figure III.14). Thus, it appears that, in the Cooper River, salinity is the most important determinant of benthic community structure, with sediment type playing a secondary role.

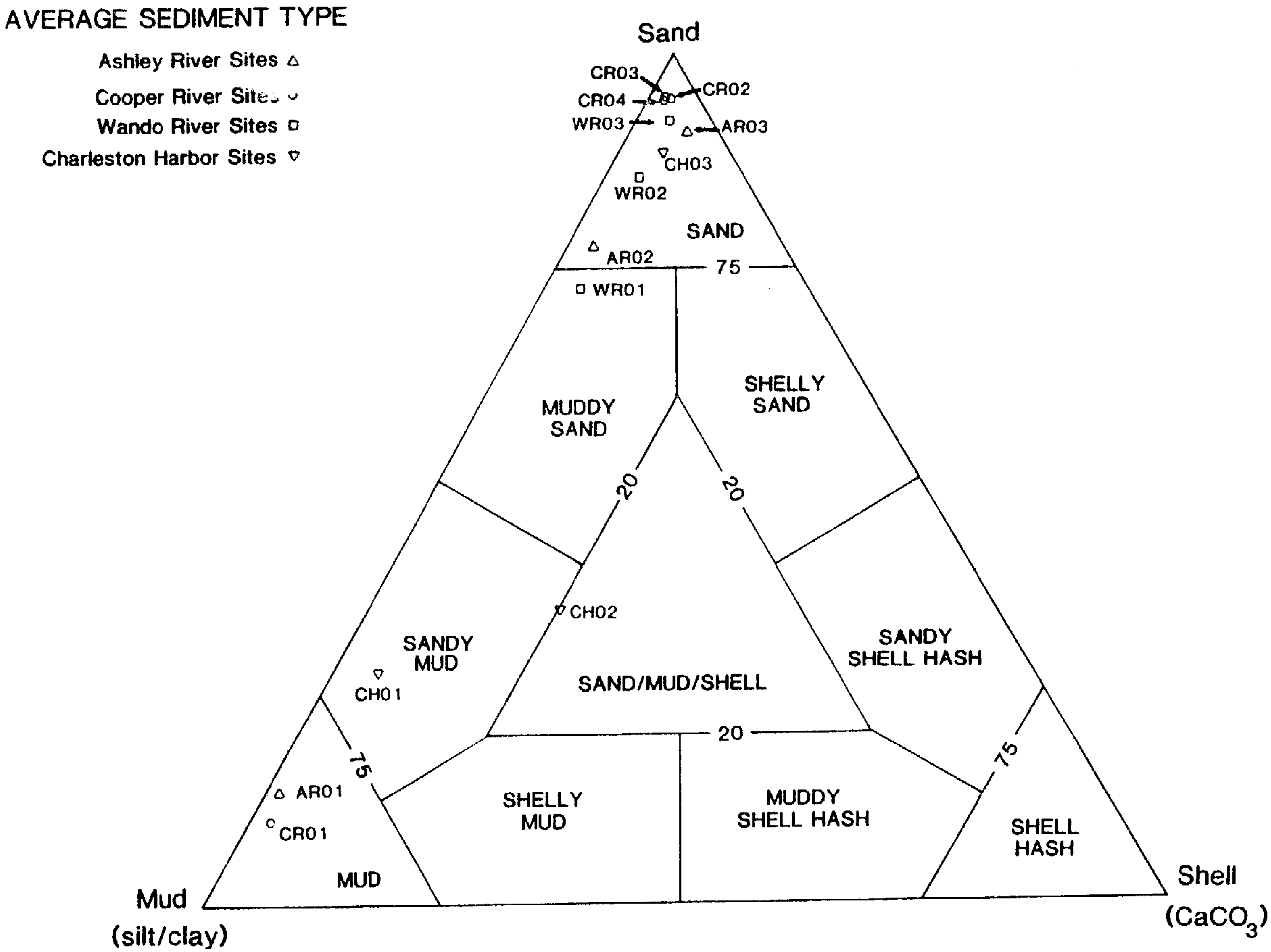


Figure VII.6. Average sediment type at grab sites based on mean percentages of sand, silt/clay and calcium carbonate. Modified from Shepard's (1954) classification scheme.

The results of the inverse cluster and nodal analyses illustrate which species groups account for the observed separation among site groups (Figure VII.5; Table VII.5). Species group A exhibited high to very high constancy at upriver sites CR04 and CR03 (site groups 1 and 2, respectively), but only moderate to very low constancy at the two downriver sites. Six of the seven taxa in this group were among the 10 most abundant taxa throughout the Cooper River (Figures VII.7a through VII.7j). All members of this group, with the possible exception of the unidentified nematodes, may be characterized as either psammophilic ("sand-loving") or low salinity (brackish-to-freshwater) species. The former category includes the amphipods *Lepidactylus dytiscus* and *Monoculodes* sp. A, while the latter includes the polychaete *Scolecolepides virides* and the amphipod *Gammarus tigrinus* (Bousfield, 1973; Croker, 1967; Roberts *et al.*, 1975). Although no information could be found on the sediment or salinity preferences of the congeneric isopods *Chiridotea almyra* and *C. stenops*, it appears from their distribution in the Cooper River that both are psammophilic species, with *C. almyra* occurring most often in tidal freshwater and in brackish water with salinities of up to 10 ppt, while *C. stenops* rarely occurred in freshwater but was frequently found in salinities ranging from 5 to 25 ppt (Figures VII.7g and VII.7h).

Species group B was characterized by its moderate constancy and fidelity among samples taken at the site furthest upriver, CR04 (site group 1). This group includes the isopod *Cyathura polita* which is largely restricted to sand bottoms and is a dominant organism in the meso- and oligohaline zones of the Chesapeake Bay (Maurer *et al.*, 1976; Roberts *et al.*, 1975). Group B also includes the oligochaetes and two families of insect larvae, the Chironomidae or "non-biting midges" and the Ceratopogonidae or "biting midges" (commonly known as "no-see-ums"). Although representatives of these taxa are found in a variety of aquatic habitats, they appear to comprise a dominant component of the soft-bottom benthos only in the oligohaline and freshwater reaches of the Charleston Harbor estuary. All three taxa are herbivores and scavengers, feeding on algae and other organic debris; however, some chironomids may be predaceous, feeding on smaller chironomid larvae or oligochaetes (Brigham *et al.*, 1982).

Species group C included only two species, the mud snail *Nassarius vibex* and the polychaete *Glycinde solitaria*. These species are most common in muddy sand and mud (Abbctt, 1968; Gardiner, 1975). Not surprisingly, then, this group was very highly faithful to the only Cooper River site characterized by muddy sediments, CR01 (site group 4).

Species group D was also characterized by its moderate fidelity to samples from station CR01. Most members of this species group, including the amphipods *Melita nitida* and *Corophium lacustre*, the polychaete *Eteone heteropoda* and the boring bivalve *Petricola pholadiformis*, commonly inhabit muddy sediments, often in association with stiff clay, shell material or other types of firm substrata such as aids to navigation, pilings and floating docks (Fox and Ruppert, 1985). All of these substrates occurred in the vicinity of station CR01.

Species group E was also highly faithful to samples from site group 4 (station CR01). Most of these species, particularly the cumaceans (*Leucon americanus, Cyclaspis varians* and *Oxyurostylis smithi*) appear to prefer high salinity (upper meso- to euhaline) habitats characterized by relatively fine-grained sediments falling within a narrow size range (Watling, 1979; Modlin and Dardeau, 1987; Abbott, 1968).

Species group F included several common estuarine polychaetes and the opportunistic bivalve *Mulinia lateralis*. These species were very highly constant but only moderately faithful to station CR01 (site group 4). It is evident from their occurrence at both muddy and sandy sites throughout the estuary, as well as from the literature, that many of these species are highly eurytopic with respect to both sediment type and salinity regime (Fox and Ruppert, 1985; Johnson, 1984; Pettibone, 1963; Stanley, 1970; Williams, 1984b; Wolff, 1973).

Although these analyses did not reveal any obvious differences related to the effects of rediversion, it appears that a few of the numerically dominant species may have undergone changes in abundance in certain reaches of the river subsequent to rediversion (Figures VII.7a through VII.7j). For example, the polychaete *Paraprionospio pinnata* has been periodically much more abundant at station CR01 in the three years following rediversion than it was in the year immediately preceding this event (Figure VII.7a). Although *P. pinnata* is generally considered to be euryhaline, based on its wide distribution in habitats ranging from mesohaline to euhaline, it may prefer the more stable polyhaline salinities characteristic of the lower estuary during post-rediversion sampling periods (Figure III.14). On the other hand, these episodes of peak abundance have been interspersed with periods of precipitous decline which are seemingly unrelated to salinity fluctuations or any other recognizable pattern of seasonal or yearly periodicity. This pattern of highly variable abundance, with increases often associated with some environmental perturbation, is characteristic of this species and other euryhaline opportunists, as well (Boesch, *et al.* 1976).

Conversely, another spionid polychaete, *Scolecolepides viridis*, was much more abundant at station CR03 during most of the year preceding rediversion than it was at the same site following this event (Figure VII.7b). There is no evidence that this decline was accompanied by a corresponding increase in abundance of this species further upriver, *i.e.*, there was no apparent shift in the distribution of this low salinity species, only a steady decline at the station where it was formerly most abundant. Considering the distance

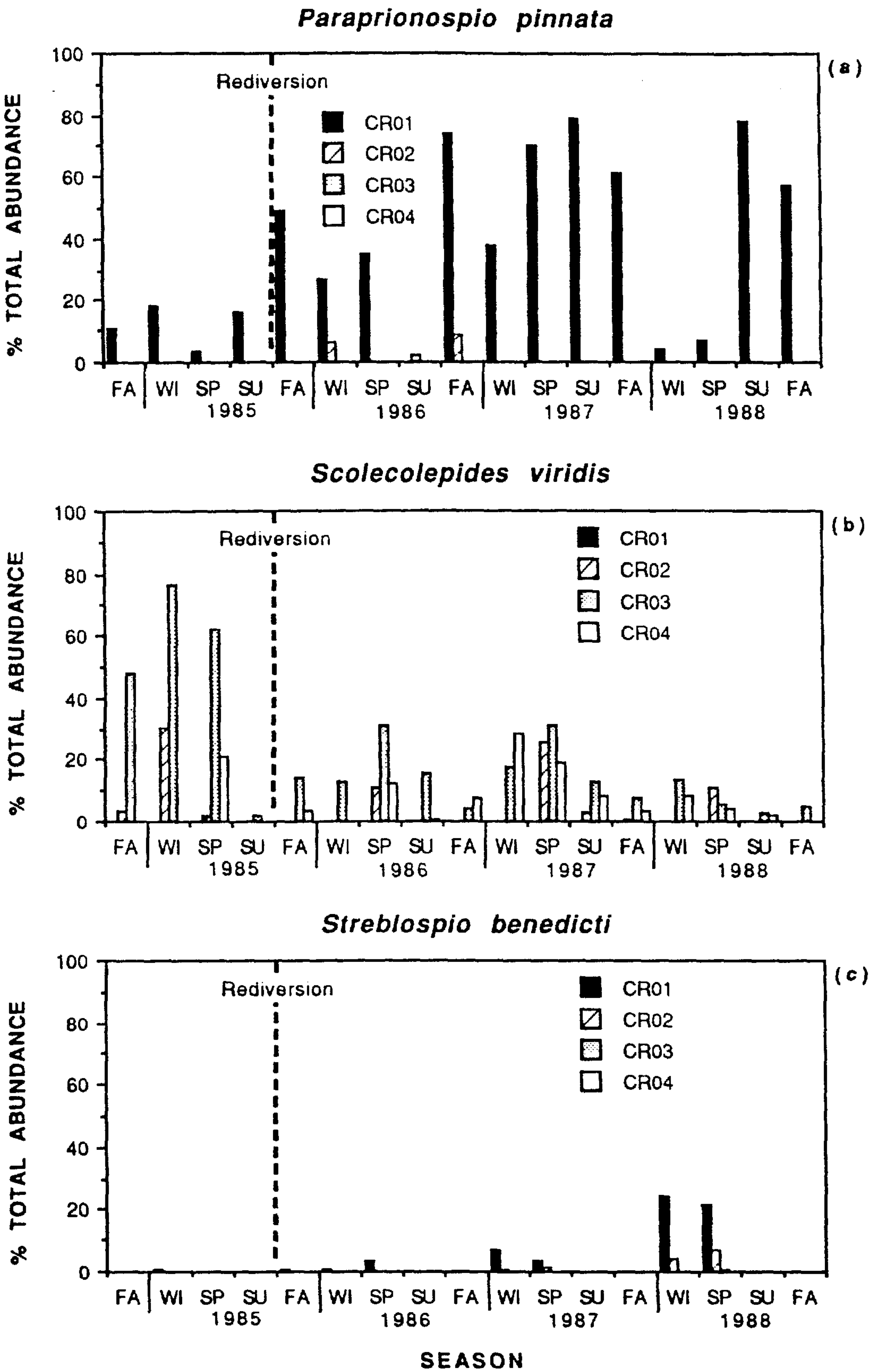


Figure VII.7. Percentage of total faunal abundance contributed by each of the 10 most abundant species to pooled replicate grab samples collected from each of the Cooper River sites during each season of the four-year study.

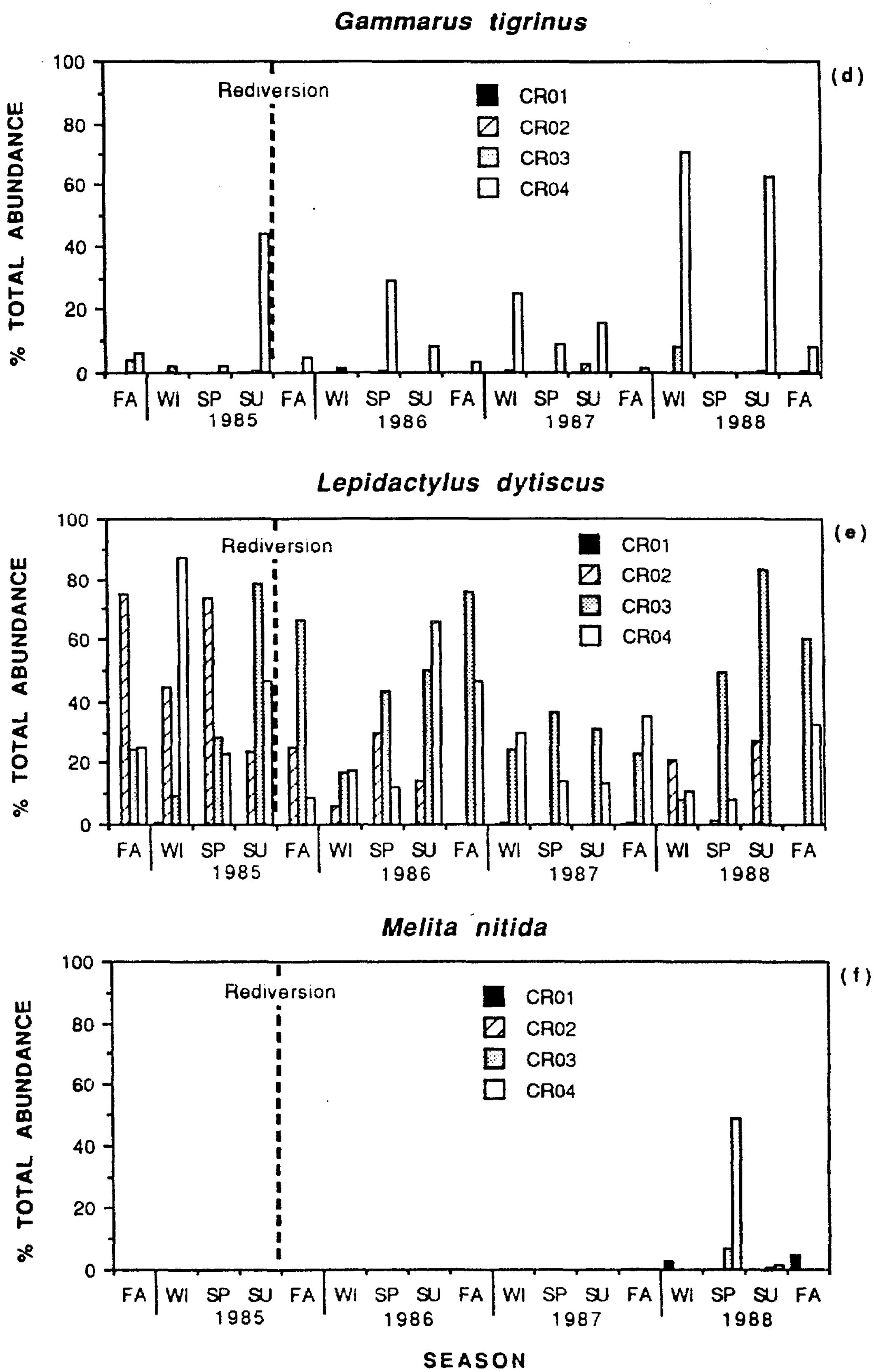
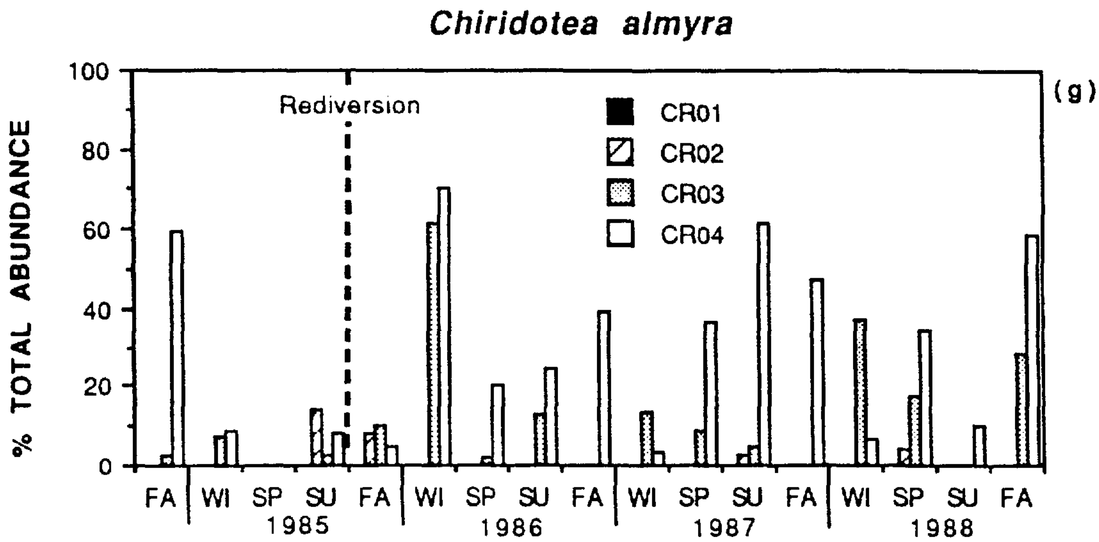
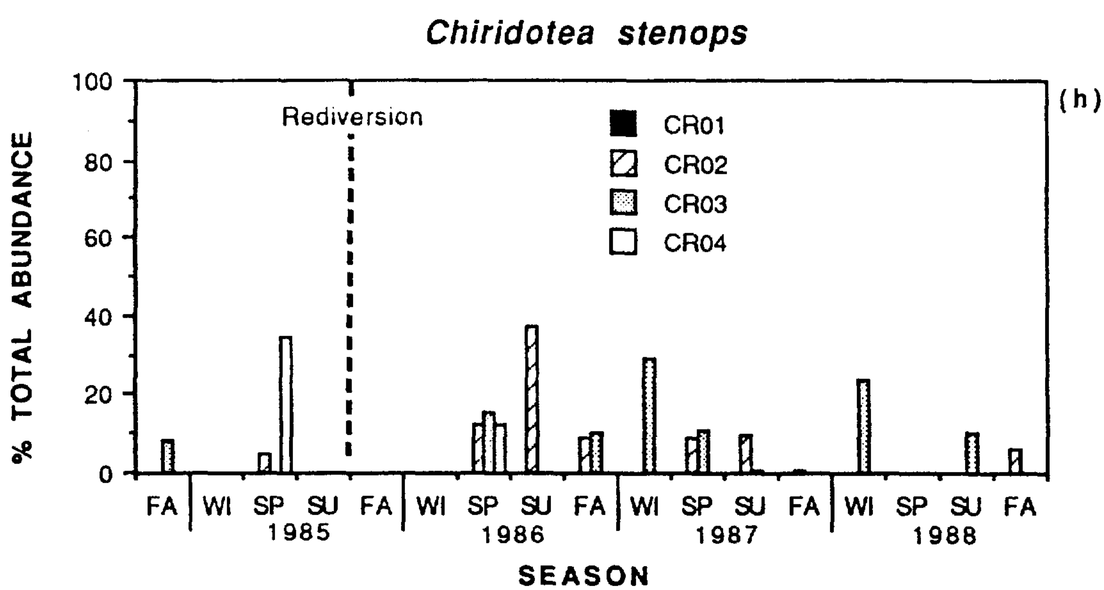


Figure VII.7. (Continued)





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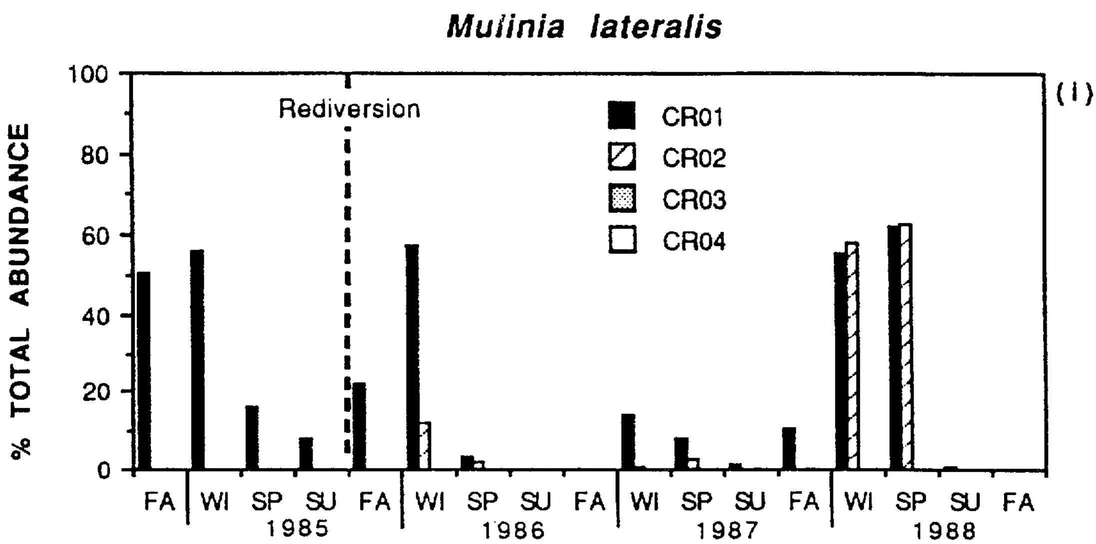


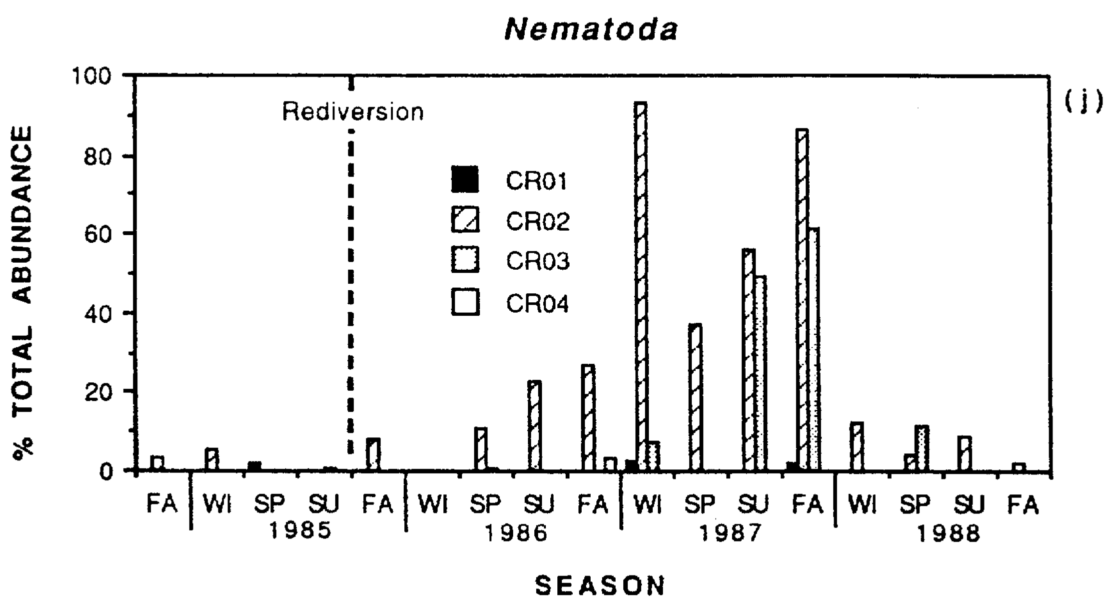


Figure VII.7 (Continued)

between stations, however, it is conceivable that such a shift occurred on a spatial scale smaller than our sampling design was capable of detecting.

The haustoriid amphipod *Lepidactylus dytiscus* underwent a similar decline in abundance at station CR02 following rediversion, although it was still periodically abundant at stations CR03 and CR04. Since this species, like *P. pinnata*, is euryhaline in its distribution and is known to thrive in euhaline habitats (Holland and Dean, 1977), it is doubtful that the modest rise in salinity at station CR02 would have affected this species adversely. Furthermore, since sediment composition, water temperature and dissolved oxygen concentrations have all remained essentially the same following rediversion, it is unlikely that any of these factors could have accounted for the observed decline in abundance of this or any other species. Alternative explanations for the apparent decline of *L. dytiscus* and *Scolecolepides viridis* at stations CR02 and CR03, respectively, include 1) atypically high abundances during the year immediately preceding rediversion, 2) biotic interactions such as competitive exclusion or predation by another species which invaded this part of the estuary subsequent to rediversion or 3) a negative response to an unmeasured environmental variable.

Wando River:

Fewer macrofaunal organisms (5,045) representing a greater number of taxa (171) were collected at the three Wando River sites than at the four Cooper River sites during the same four-year period. Mean abundances ranged from a low of 2.33 individuals/grab at station WR02 in Fall, 1988 to a high of 167.00 individuals/grab at station WR02 in Spring, 1986. As in the Cooper River, there were no consistent trends in the mean abundance of macrofauna with respect to site, season or year of collection; nor were there any obvious differences between pre-rediversion and post-rediversion sampling periods (Figure VII.8).

Once again, these observations are supported by the results of a three-way analysis of variance in which all three sources of variation tested (site, season and year) exhibited highly significant second-order interaction effects (Tables V.1 and V.2). Thus, although mean abundances were significantly higher in 1986 overall, this was largely due to a dramatic increase in the number of nematodes at one station (WR02) during one season (spring). Similarly, mean abundances were significantly different among all three sampling sites, with WR01 having the highest and WR02 having the lowest overall mean abundances. These differences were highly dependent on the year and season of collection, however.

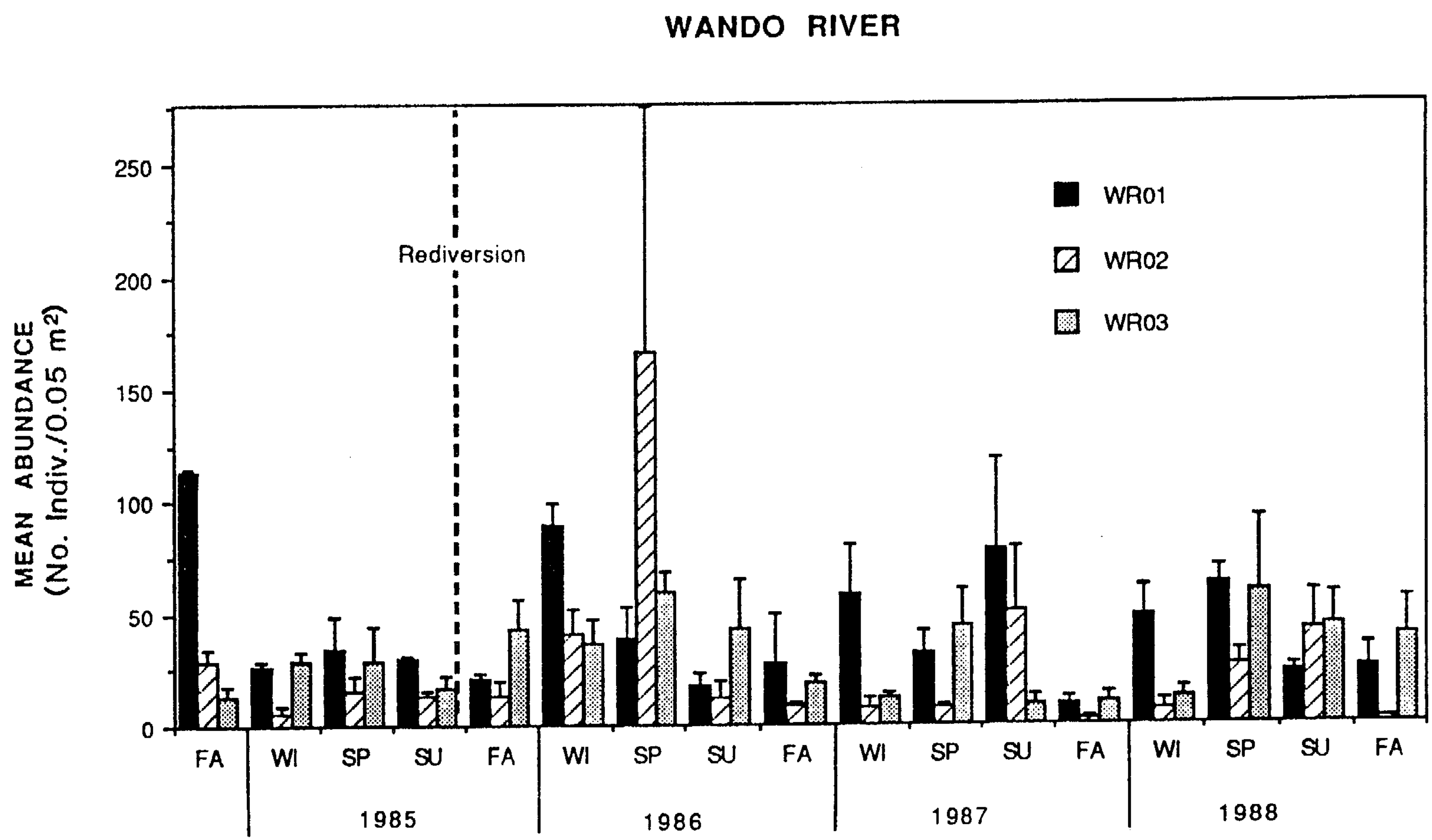


Figure VII.8. Mean abundance of macrofaunal organisms collected at each of the Wando River sampling sites during each season of the four-year study. (Vertical lines represent one standard error of the mean).

The results of species diversity analyses were somewhat less equivocal. As in the Cooper River, diversity (H') values in the Wando ranged widely, from a low of 0.57 bits/individual at station WR03 in Fall, 1988 to a high of 4.38 bits/individual at station WR01 in Winter, 1987 (Figure VII.9a). Unlike the Cooper River, however, these values were, with one exception, consistently higher at the station furthest downriver (WR01) than at either of the other two sites, and were generally higher at WR02 than at WR03, the station furthest upriver. The increase in diversity toward the mouth of the Wando River was due, primarily, to a corresponding increase in species richness (Figure VII.9c).

A three-way analysis of variance comparing the total number of unique species in pooled replicate grab samples among sites, seasons, and years showed that all three sources of variation exhibited significant main effects without any significant interactions (Table VII.3). As indicated by the diversity analyses, station WR01 had a greater number of species than either of the other two sites, regardless of year or season. Despite some significant differences among certain years and seasons (Table VII.4), there were no consistent trends with respect to rediversion.

The results of a normal cluster analysis of Wando River grab data were not as sharply defined by salinity regime or sediment type as they were in the Cooper River (Figure VII.10). Although grab samples clustered primarily by site, rather than season or year of collection, there was a greater degree of overlap among samples from different sites than observed in the Cooper River. This was particularly true of site group 3 which contained samples from all three Wando River sites. These results are not surprising considering the similarity among sites with respect to both salinity and sediment type (Figures III.14 and VII.6).

Although there were no discrete groups of pre-rediversion versus post-rediversion samples, collections taken prior to rediversion at a particular station generally clustered closely together within the same site group. However, the results of our inverse cluster and nodal analyses provide no indication of which species accounted for the close similarity among pre-rediversion grab samples from the same site. The five species groups generated by an inverse analysis of Wando River grab samples were almost invariably characterized by their moderate to low constancy and fidelity among all site groups, indicating that most species were neither restricted to, nor did they consistently occur in, collections from any one site group (Figure VII.11, Table VII.6).

The single exception to this trend was the high fidelity exhibited by species group C among collections from station WR01 in site group 4. This group included several meso-

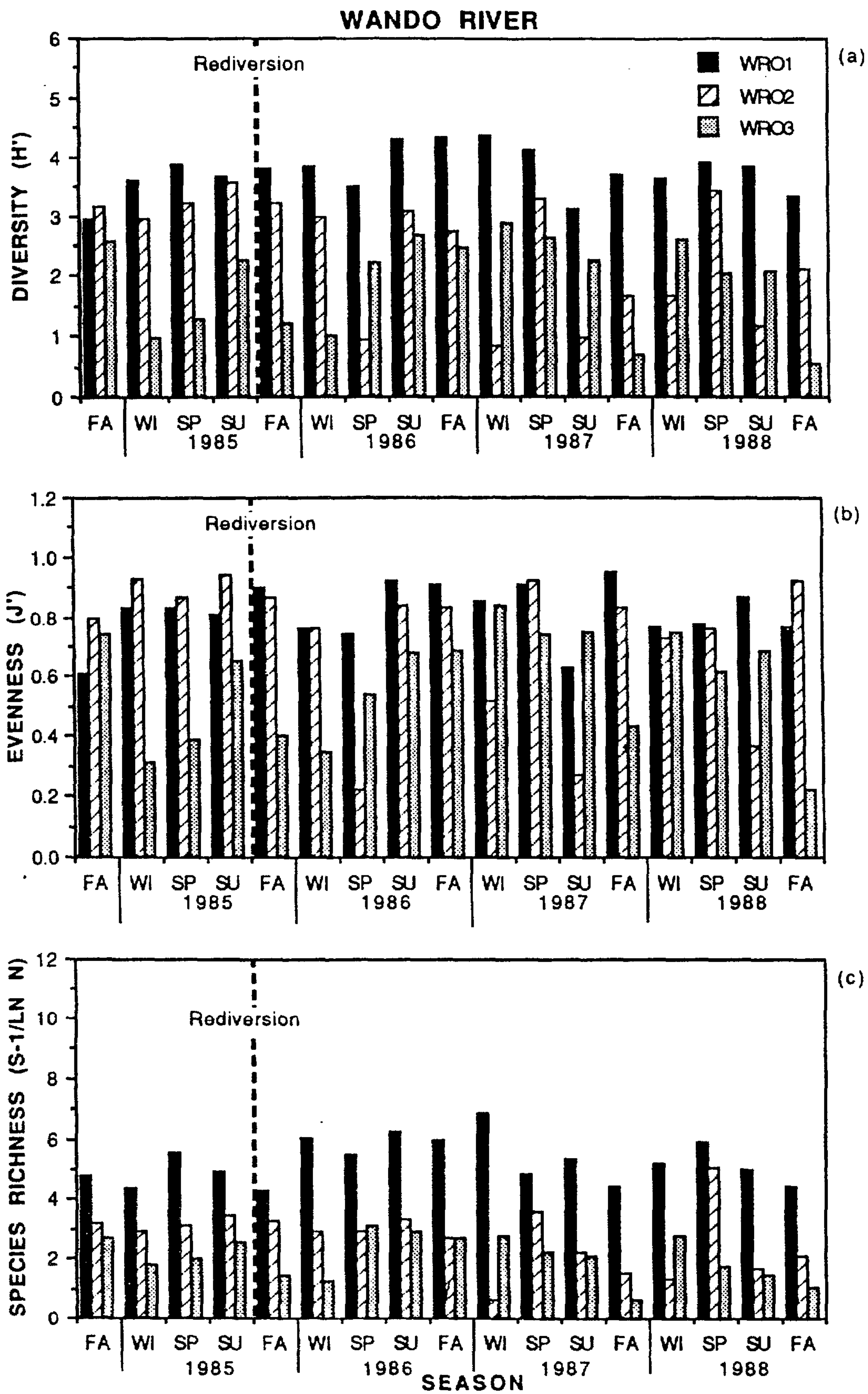


Figure VII.9. Species diversity (H'), evenness (J') and richness (S-1/ln N) values for benthic macrofaunal samples collected at each of the Wando River stations during each season of the four-year study.

Wando River

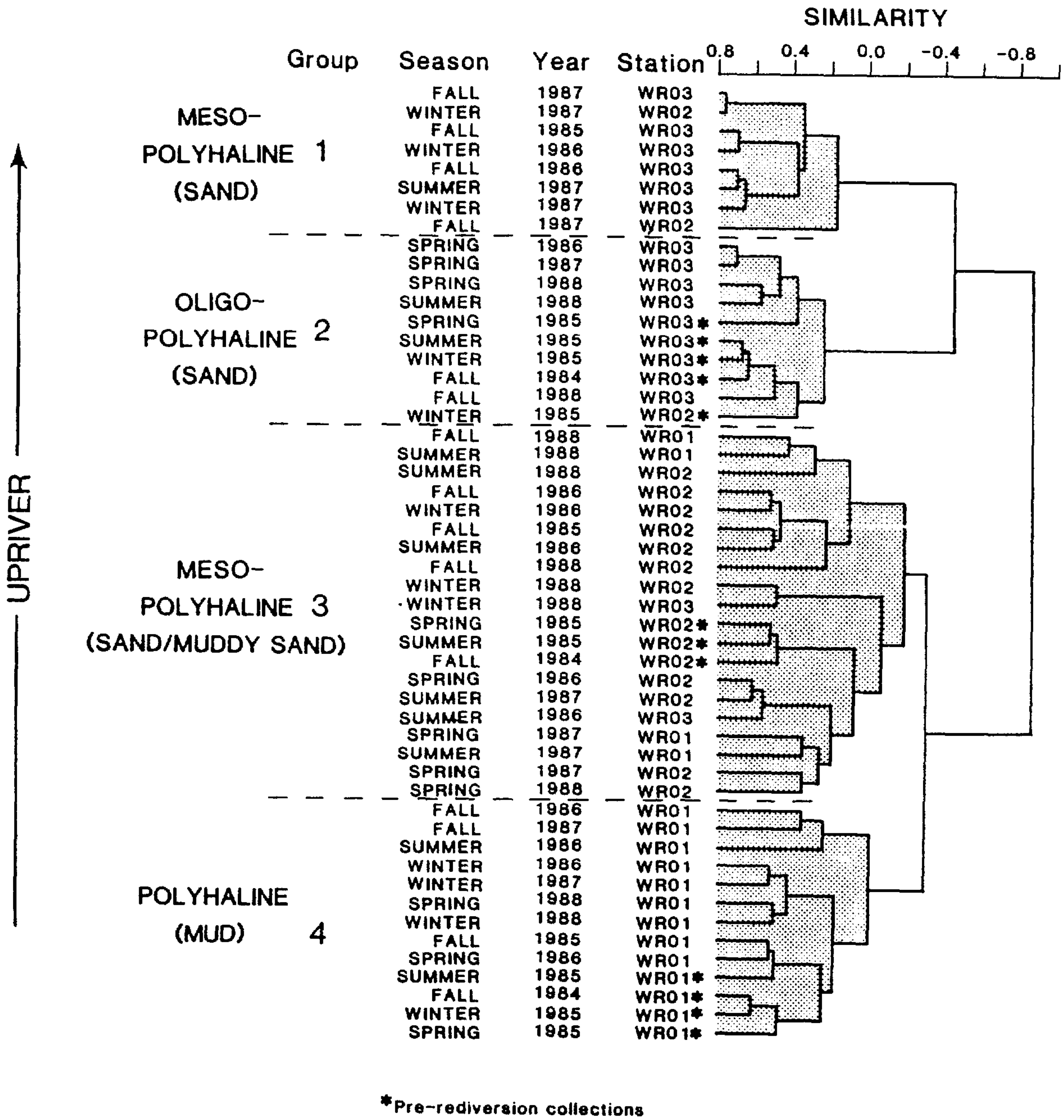


Figure VII.10. Hierarchical classification of Wando River grab samples generated by a normal cluster analysis.

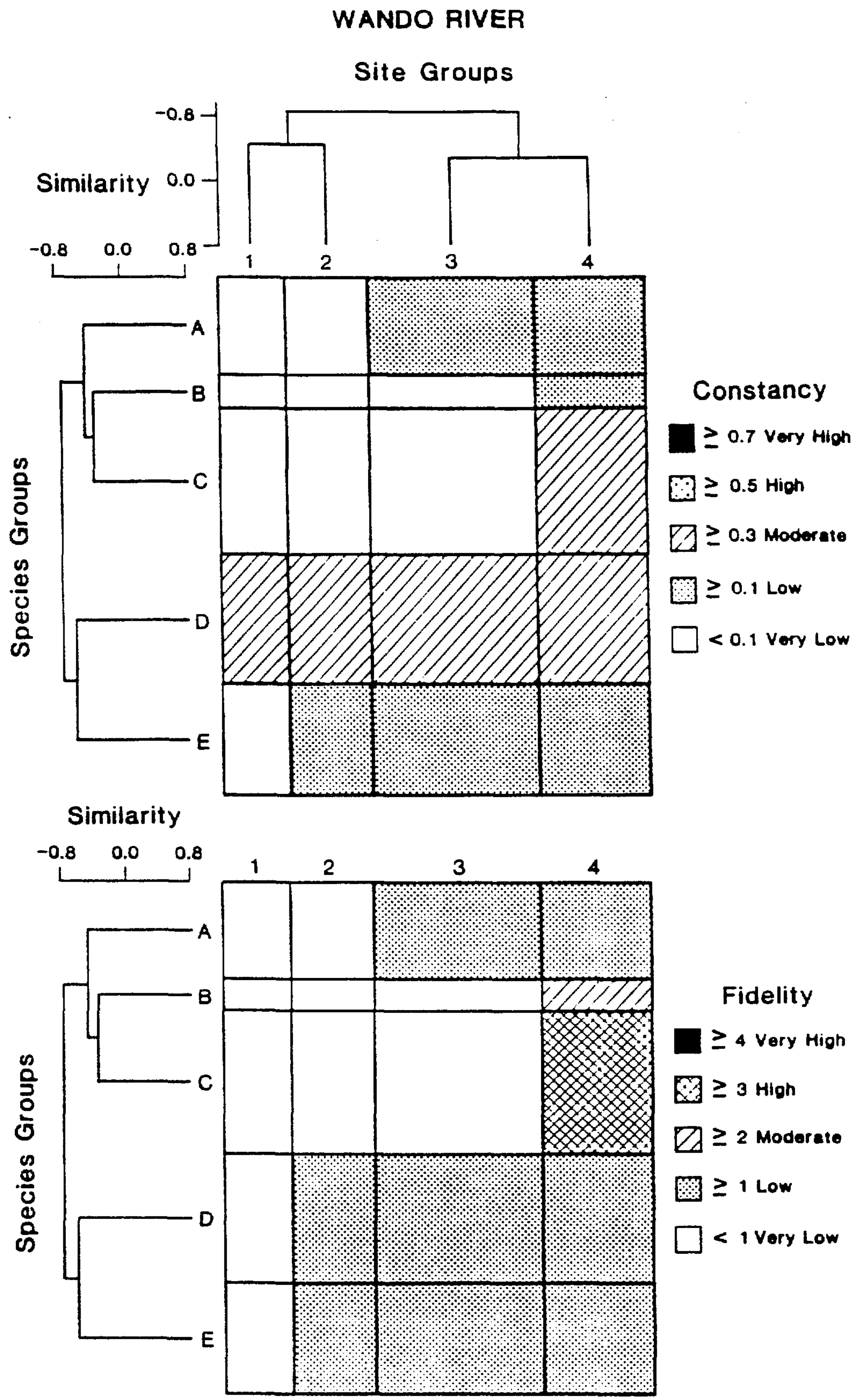


Figure VII.11. Nodal constancy and fidelity diagrams illustrating coincidences between species groups and site groups generated by inverse and normal cluster analyses of Wando River grab data.

Table VII.6. Species groups generated by an inverse cluster analysis of Wando River grab samples. (Am = amphipod; As = ascidian; B = bivalve; Cu = cumacean; D = decapod; G = gastropod; Is = isopod; My = mysid; N = nemertinean; P = polychaete).

GROUP A

Clymenella torquata (P)
Aligena elevata (B)
Oxyurostylis smithi (Cu)
Tellina sybaritica (B)
Glycera americana (P)
Nassarius vibex (G)
Carinomella lactea (N)
Melita nitida (Am)
Spiochaetopterus oculatus (P)
Paracaprella tenuis (Am)
Phoronida
Oligochaeta

GROUP B

Sabellaria vulgaris (P)
Diopatra cuprea (P)
Batea catharinensis (Am)
Lepidonotus sublevis (P)

GROUP C

Erichthonius brasiliensis (Am)
Parapleustes aestuaruis (Am)
Corophium lacustre (Am)
Molgula manhattensis (As)
Polycarpa sp. (As)
Listriella clymenellae (Am)
Unicola serrata (Am)
Anadara transversa (B)
Nucula proxima (B)
Astyris lunata (G)
Caprella equilibra (Am)
Cyathura burbancki (Is)
Petricola pholadiformis (B)
Brachidontes exustus (B)
Acteocina canaliculata (G)
Mercenaria mercenaria (B)
Tellina probrina (B)
Cistenides gouldii (P)

GROUP D

Lepidactylus dytiscus (Am)
Leitoscoloplos fragilis (P)
Eteone heteropoda (P)
Mancocuma sp. (Cu)
Edotea montosa (Is)
Monoculodes sp. A (Am)
Mulinia lateralis (B)
Streblospio benedicti (P)
Nereis succinea (P)
Nemertinea
Heteromastus filiformis (P)
Scoloplos rubra (P)
Glycinde solitaria (P)
Ostracod B
Paraprionospio pinnata (P)
Nematoda

GROUP E

Lyonsia hyalina (B)
Mediomastus californiensis (P)
Gastropoda
Trachypenaeus constrictus (D)
Glycera sp. A (P)
Leitoscoloplos robustus (P)
Leucon americanus (Cu)
Cyathura polita (Is)
Capitella capitata (P)
Chiridotea almyra (Is)
Scolecolepides viridis (P)
Tellina texana (B)
Cyclaspis varians (Cu)
Neomysis americana (My)

to euhaline species frequently found in muddy sand or mud (e.g., the bivalves, *Nucula proxima* and *Mercenaria mercenaria*; the gastropod *Acteocina canaliculata*; and the isopod *Cyathura burbancki*) as well as numerous species commonly associated with oyster shell, rocks, pilings, floating docks, aids to navigation, or other types of firm substrata (Abbott, 1968; Bousfield, 1973; Fox and Ruppert, 1985). This latter group included both free-swimming and tubicolous amphipods (*Parapleustes aestuarius, Caprella equilibra, Erichthonius brasiliensis* and *Corophium lacustre*) as well as sessile ascidians (*Molgula manhattensis* and *Polycarpa* sp.) and bivalves (*Anadara transversa, Brachidontes exustus*, and *Petricola pholadiformis*). The scorched mussel, *B. exustus*, was closely associated with the false angel wing, *P. pholadiformis*, which is often found boring in stiff clay or peat (Abbott, 1968). A commensal relationship between members of the two families of bivalves represented by *B. exustus* and *P. pholadiformis* (Mytilidae and Petricolidae) has also been documented in the literature (Abbott, 1968; Wolff, 1973). The substantial clay content of sediments at station WR01 may account for the presence of these and perhaps other species in group C. Other sources of firm substrata in the vicinity of WR01 include channel markers and rip-rap along the shore of a diked disposal area on Daniel Island.

The only other species group that merits discussion is species group D, which included many of the most abundant and ubiquitous macrofaunal species found throughout the estuary. This is reflected in their moderate constancy among collections belonging to all Wando River site groups.

Although the cluster and nodal analyses provide little evidence of rediversion effects on macrobenthic community structure in the Wando River, there were some changes in the relative abundance of numerically dominant species subsequent to rediversion (Figures VII.12a through VII.12j). As in the Cooper River, the polychaete *Paraprionospio pinnata* became periodically much more abundant in the Wando River during the three years following rediversion, particularly at station WR02 and, to a lesser extent, at WR01 as well (Figure VII.12a). The stations at which *P. pinnata* was most abundant in the two rivers (CR01 and WR02) had salinities that were consistently in the poly- to euhaline range following rediversion but had vastly different sediment types, with CR01 sediments being composed of >85% silt/clay on average and WR02 sediments being composed of >85% sand (Figure VII.6). These results suggest that *P. pinnata* may be more eurytopic with respect to sediment type than it is with respect to salinity.

Another group of organisms that increased markedly in number following rediversion (particularly at stations WR02 and WR03) were the nematodes (Figure VII.12j). The ecological significance of this increase is uncertain in the absence of any knowledge regarding the specific identify and, consequently, the life history attributes or habitat

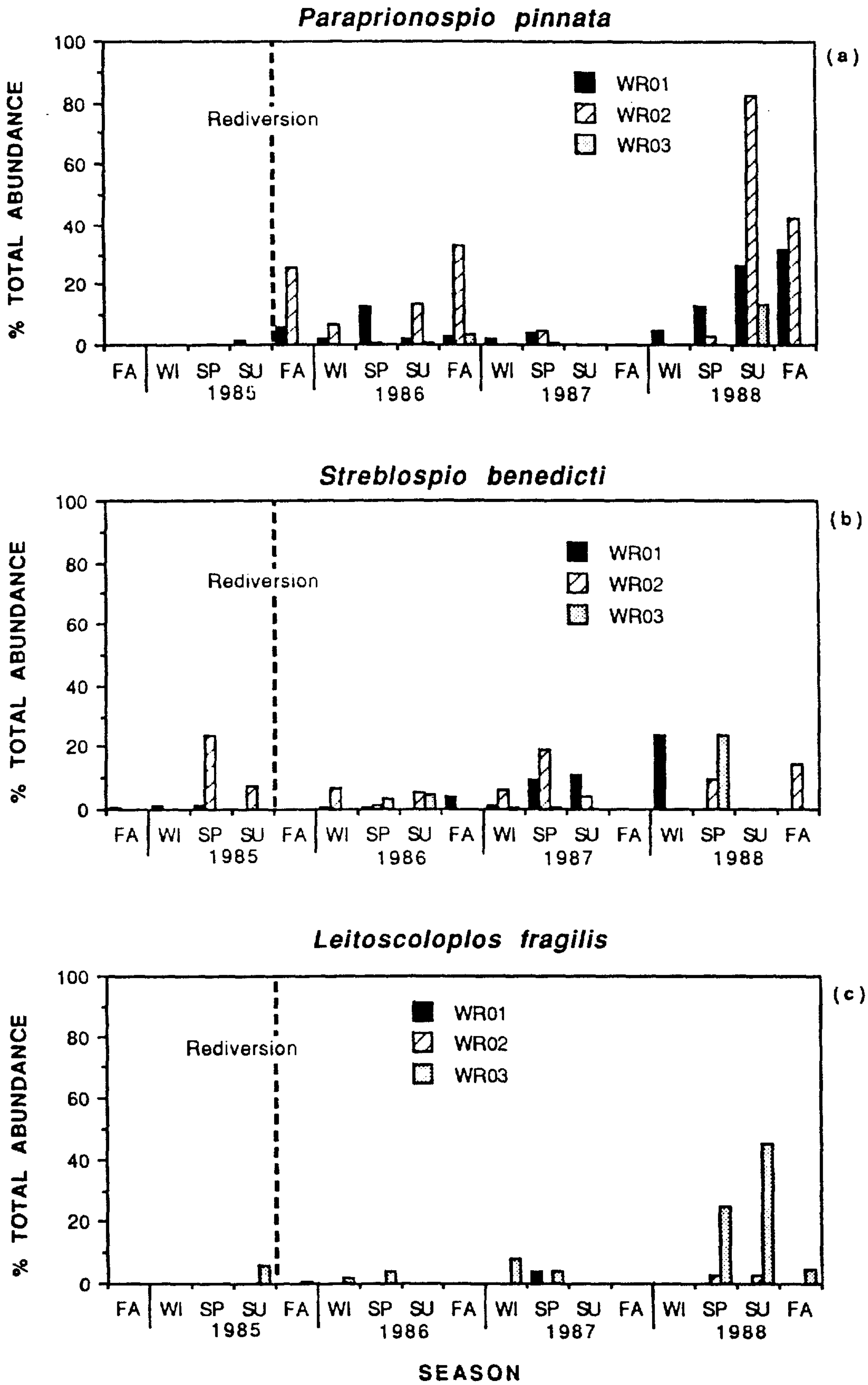
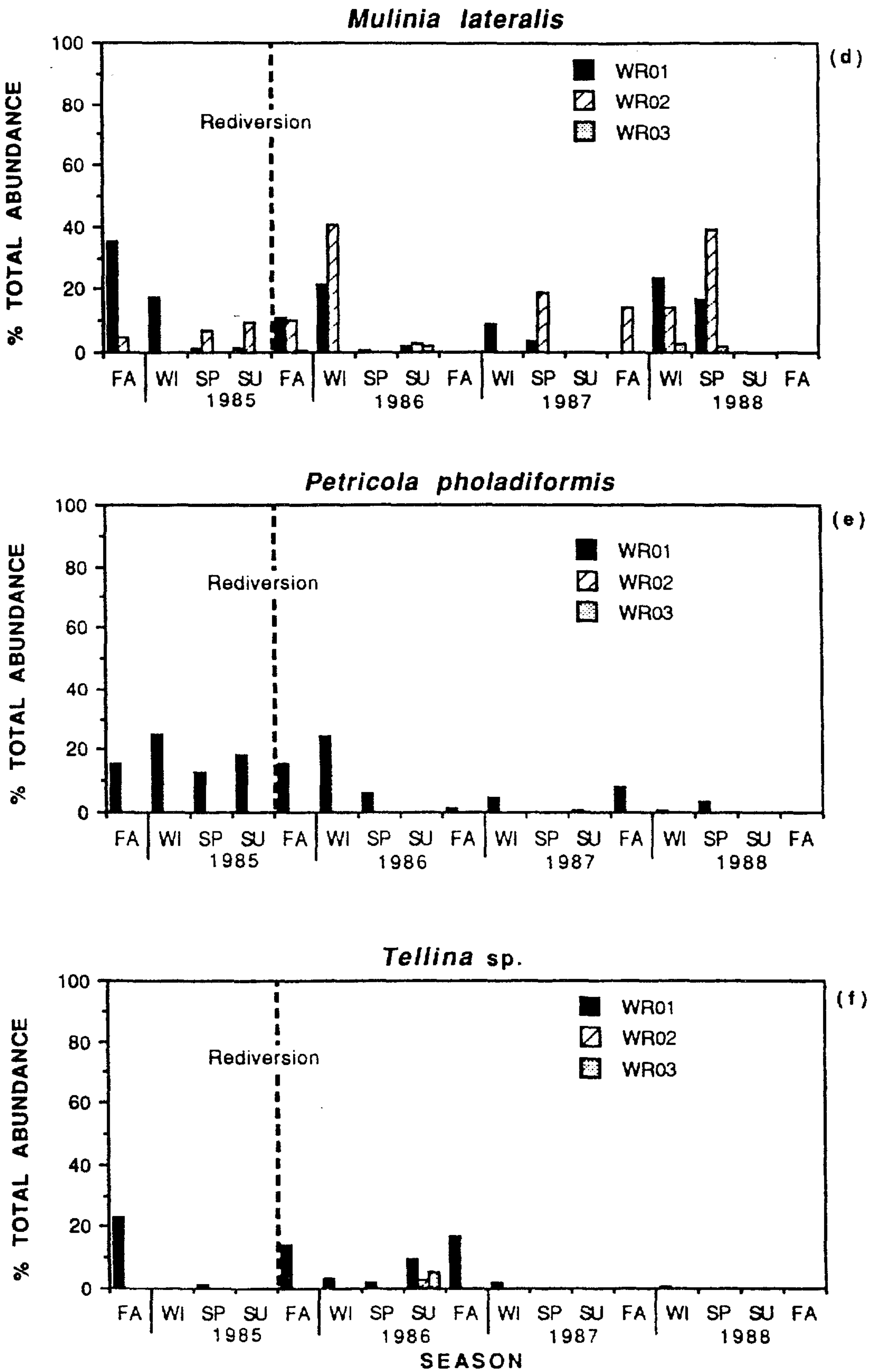


Figure VII.12. Percentage of total faunal abundance contributed by each of the 10 most abundant species in pooled replicate grab samples collected from each of the Wando River sites during each season of the four-year study.



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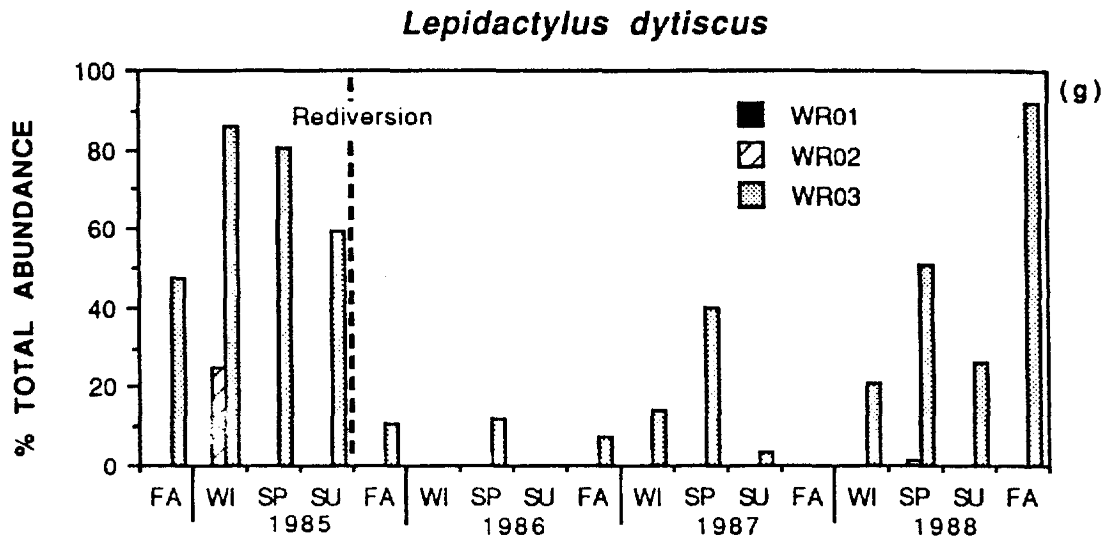

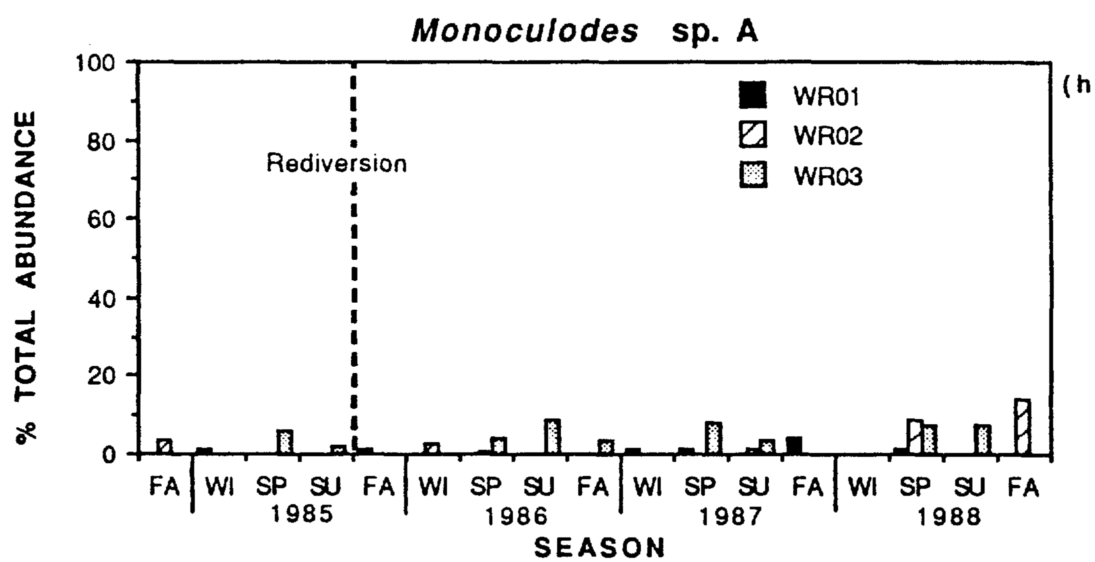

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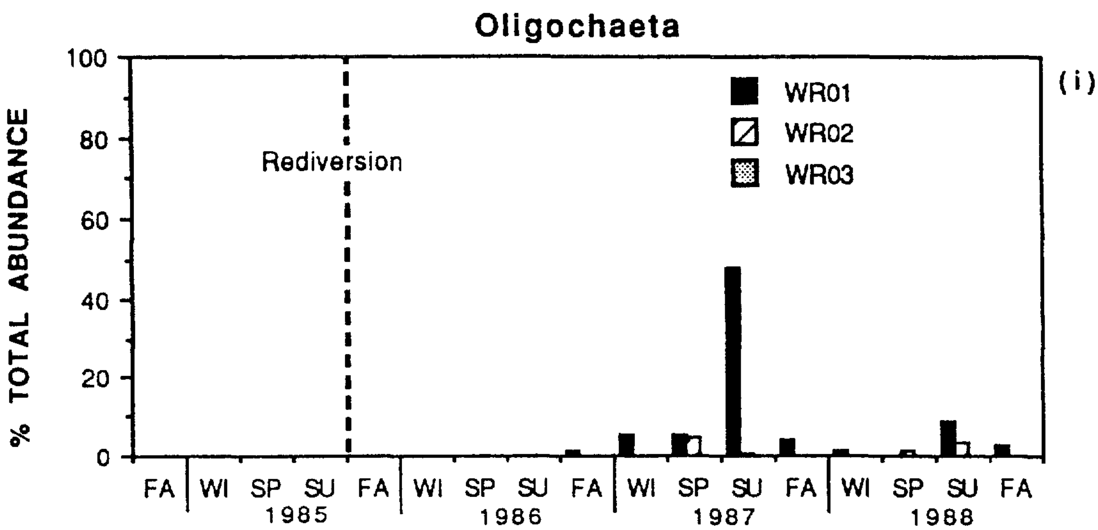


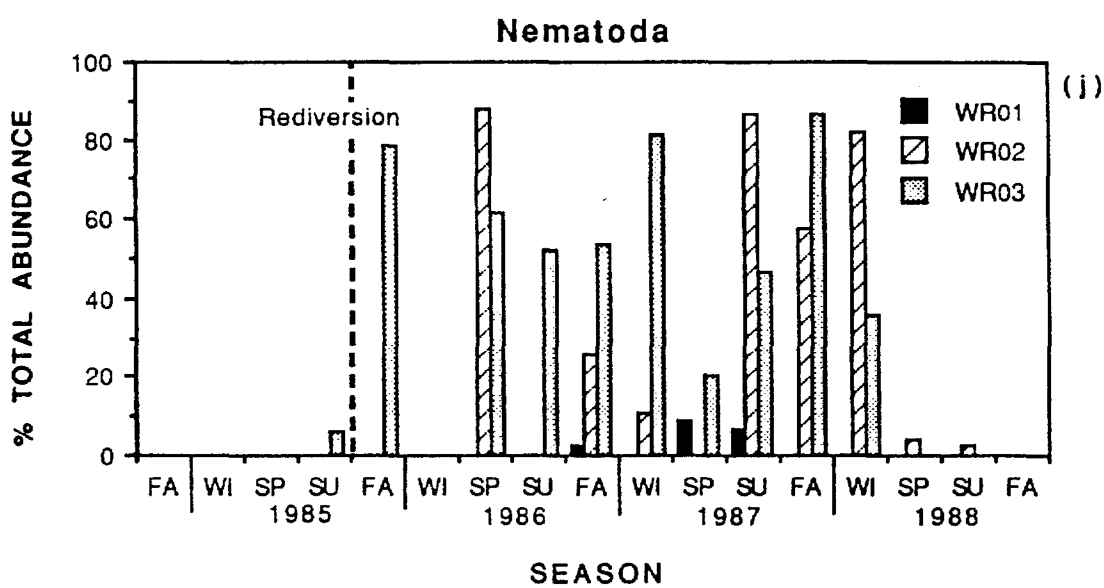


Figure VII.12. (Continued)

requirements of these organisms. As a group, nematodes are ubiquitous, represent all feeding strategies, and have been described as "perhaps the most highly adaptable of all the metazoan phyla" (Pennak, 1978). They are also able to withstand low concentrations of dissolved oxygen for extended periods of time; however, the absence of any obvious decline in dissolved oxygen concentrations in the Wando River subsequent to rediversion makes it unlikely that this particular physiological capability would have afforded the nematodes any competitive advantage over other macrofaunal organisms.

Species that declined in abundance after rediversion included the haustoriid amphipod *Lepidactylus dytiscus* and the bivalve *Petricola pholadiformis* (Figures VII.12g and VII.12e). *Lepidactylus dytiscus* occurred almost exclusively, both before and after rediversion, at the one station where sediments had the highest sand and lowest silt/clay content, WR03. This finding is consistent with observations made by other researchers on the distribution of *L. dytiscus* in intertidal habitats where it was restricted to sediments having <3% silt/clay (Holland and Dean, 1977). As in the Cooper River, the decline in abundance of this species does not bear any clear relationship to changes in salinity or sediment composition, both of which remained essentially the same after an initial increase in salinity at two of the three sites following the first winter sampling period in 1985 (Figure III.14). The pattern is further confused by a striking increase in abundance of *L. dytiscus* to pre-rediversion levels in Fall, 1988. High abundances of *L. dytiscus* coincided with low abundances of nematodes, and *vice versa*. This suggests that these animals may be affecting one another's distribution either directly or indirectly, through competition, predation or physical disturbance of the sediments. Alternatively, they may be responding in opposite ways to the same environmental stimulus. In either case, the specific role (if any) played by rediversion in effecting these changes remains uncertain.

Charleston Harbor:

During the four-year study, the greatest number of macrofaunal organisms (21,650) and the greatest number of species (252) were collected at the three sites in the Charleston Harbor basin. Mean abundances ranged from a low of 11.00 individuals/grab at station CH01 in Summer, 1985 to a high of 1626.33 individuals/grab at station CH02 in Summer, 1988. As in the Cooper and Wando Rivers, there were no apparent trends in the mean abundance of organisms with respect to site, season or year of collection (Figure VII.13). Nor were there any consistent differences between pre- and post-rediversion sampling periods.

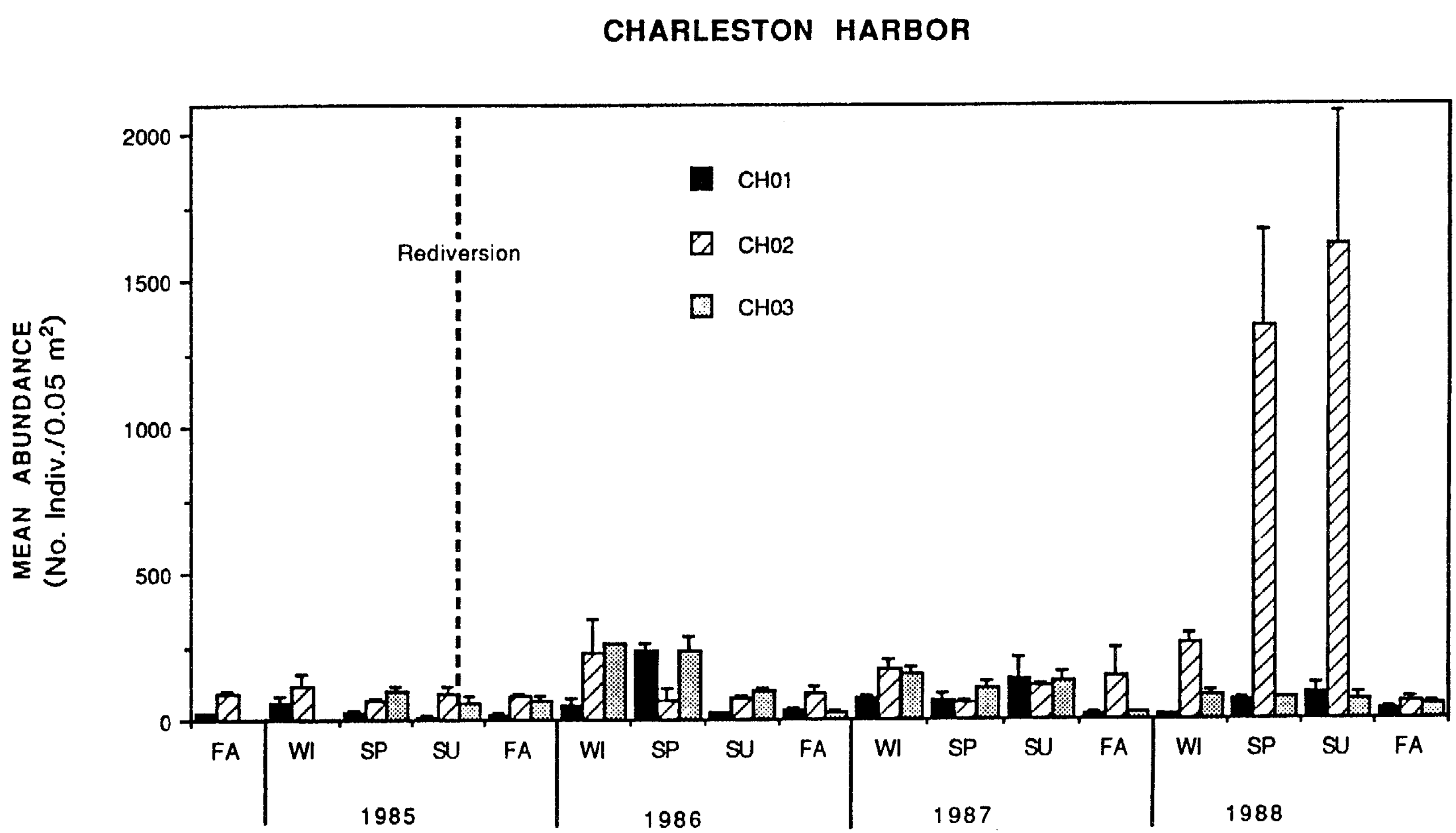


Figure VII.13. Mean abundance of macrofaunal organisms collected at each of the Charleston Harbor sampling sites during each season of the four-year study. (Vertical lines represent one standard error of the mean).

These observations are supported by the results of a three-way analysis of variance in which all three sources of variation tested (site, season and year of collection) exhibited highly significant first- and second-order interaction effects (Table VII.1). Statistical tests of significance were performed only for stations CH01 and CH02 since grab samples were not taken at CH03 during the first two pre-rediversion sampling periods. The results of *a posteriori* comparisons of group means indicated that total abundances were significantly greater at station CH02 than at CH01 (Table VII.2); however, this disparity was primarily a function of the dramatic increase in abundance of a small opportunistic bivalve, *Mulinia lateralis*, at station CH02 in the spring and summer of 1988.

Species diversity (H') values ranged from a low of 0.70 bits/individual at station CH02 in Summer, 1988 to a high of 4.70 bits/individual at station CH03 in Spring, 1985 (Figure VII.14). As in the Cooper and Wando Rivers, there were no consistent differences in species diversity, evenness or richness values among sites, seasons, or years of collection; although, both species diversity and species richness were more frequently higher at station CH03 than at either of the other two sites. There were no apparent differences in any of these indices between pre- and post-rediversion sampling periods.

A three-way analysis of variance comparing the total number of unique species collected at stations CH01 and CH02 among all sampling periods showed both site and season, but not year of collection, to have significant main effects with no significant interactions (Table VII.3). The results of *a posteriori* comparisons among group means showed that the total number of species collected was significantly greater in summer than in fall and was also significantly greater at station CH02 than at CH01 (Table VII.4). Tests of significance were not applied to station CH03 since missing data and, consequently, unequal cell sizes would have produced erroneous results. Nevertheless, the total number of species was, in fact, greater at station CH03 than at either of the other two sites during 12 of the 15 sampling periods in which all three sites were sampled.

The higher species diversity at station CH03 may reflect the site's more stable salinity regime, which was consistently in the high polyhaline to euhaline range throughout the study (Figure III.14). Similarly, salinities were less variable at station CH02 than at CH01, perhaps accounting, at least in part, for the disparity in species number between these two sites as well. More importantly, however, sediments differed markedly among all three harbor basin sites (Figure VII.6). Thus, unlike the sediments at stations CH01 and CH02 which had a large silt/clay component, the predominantly sandy sediments at station CH03 provided the only suitable substrate for a number of psammophilic species. Similarly, the relatively high percentage of shell hash at station CH02 may have provided the only suitable substrate for a variety of sessile and motile epifaunal species.

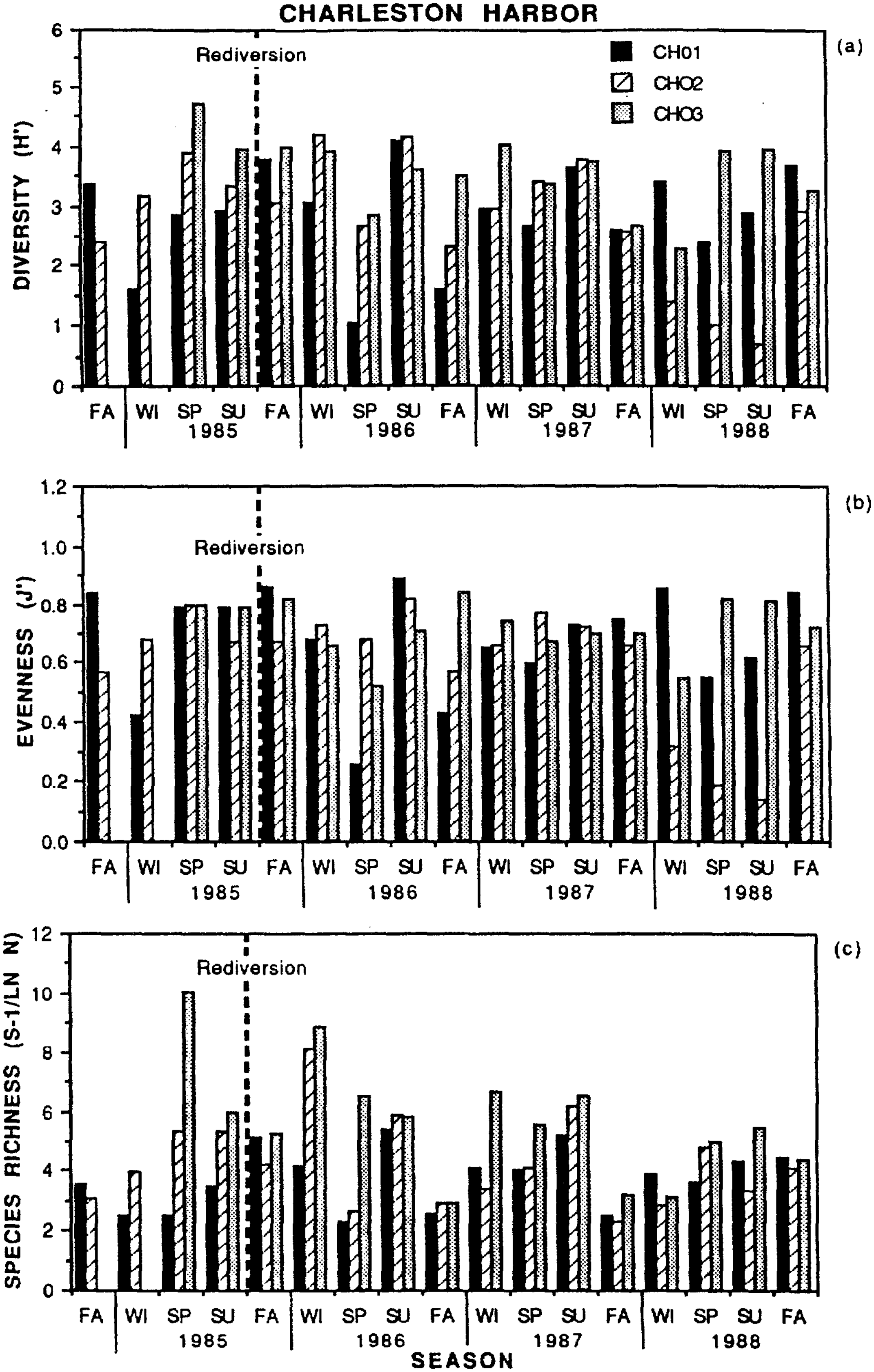


Figure VII.14. Species diversity (H'), evenness (J') and richness (S-1/ln N) values for benthic macrofaunal samples collected at each of the Charleston Harbor stations during each season of the four-year study.

The importance of sediment type in determining community composition is supported by the results of cluster and nodal analyses, as well. A normal cluster analysis generated three site groups, each consisting almost entirely of grab collections from one of the three harbor basin sites (Figure VII.15). Once again, there were no obvious patterns of similarity with respect to season or year of collection, however.

Inverse cluster and nodal analyses clearly showed which species groups accounted for the observed dissimilarity among sites (Figure VII.16, Table VII.7). Species group A, was the only group that was consistently found in collections from all three site groups. This species group contained many of the same species found in the Cooper and Wando Rivers, as well. These included the ubiquitous polychaetes *Paraprionospio pinnata, Heteromastus filiformis, Mediomastus californiensis* and *Glycinde solitaria*, as well as nemertineans, oligochaetes, and bivalves (specifically, *Mulinia lateralis* and *Petricola pholadiformis*).

Species group B was characterized by its moderate constancy among collections from station CH02 (site group 2) and included a number of sessile and motile epifaunal species commonly found in muddy sand or mud mixed with gravel, shell hash, or other hard substrata (Bousfield, 1973; Caine, 1978; Fox and Ruppert, 1985; Gathof, 1984a and b; Gardiner, 1975; Pettibone, 1963; Richardson, 1971). Among these were the polychaetes *Ancistrosyllis jonesi, Bhawania heteroseta, Diopatra cuprea, Sabellaria vulgaris, Nereis succinea* and *Sigambra tentaculata*, as well as the amphipods *Paracaprella tenuis* and *Batea catharinensis*. The more frequent occurrence of these species in collections from site group 2 may reflect the somewhat higher percentage of shell hash in sediments at station CH02 compared to the other two sites.

Species group C exhibited low to very low constancy and fidelity among all site groups. Members of this group were found less frequently in collections from site group 3 (station CH01) than they were in collections from the other two site groups. Group C included the phoronids and the polychaete *Cistenides gouldii*, both of which construct tubes from sand gains. Phoronids are surface deposit feeders, while *C. gouldii* is a sub-surface deposit feeder known to prefer large mineral grains encrusted with organic debris (Whitlach, 1974). Their less frequent occurrence at the muddiest of the three Charleston Harbor sites (CH01) suggests that sediment grain size composition may be the most important factor determining the distributions of species in this group.

Species group D consisted of a large and diverse group of species, almost all of which have been cited by Fox and Ruppert (1985) as perennial inhabitants of protected beaches, creeks and sounds in South Carolina. Several of these species are found more

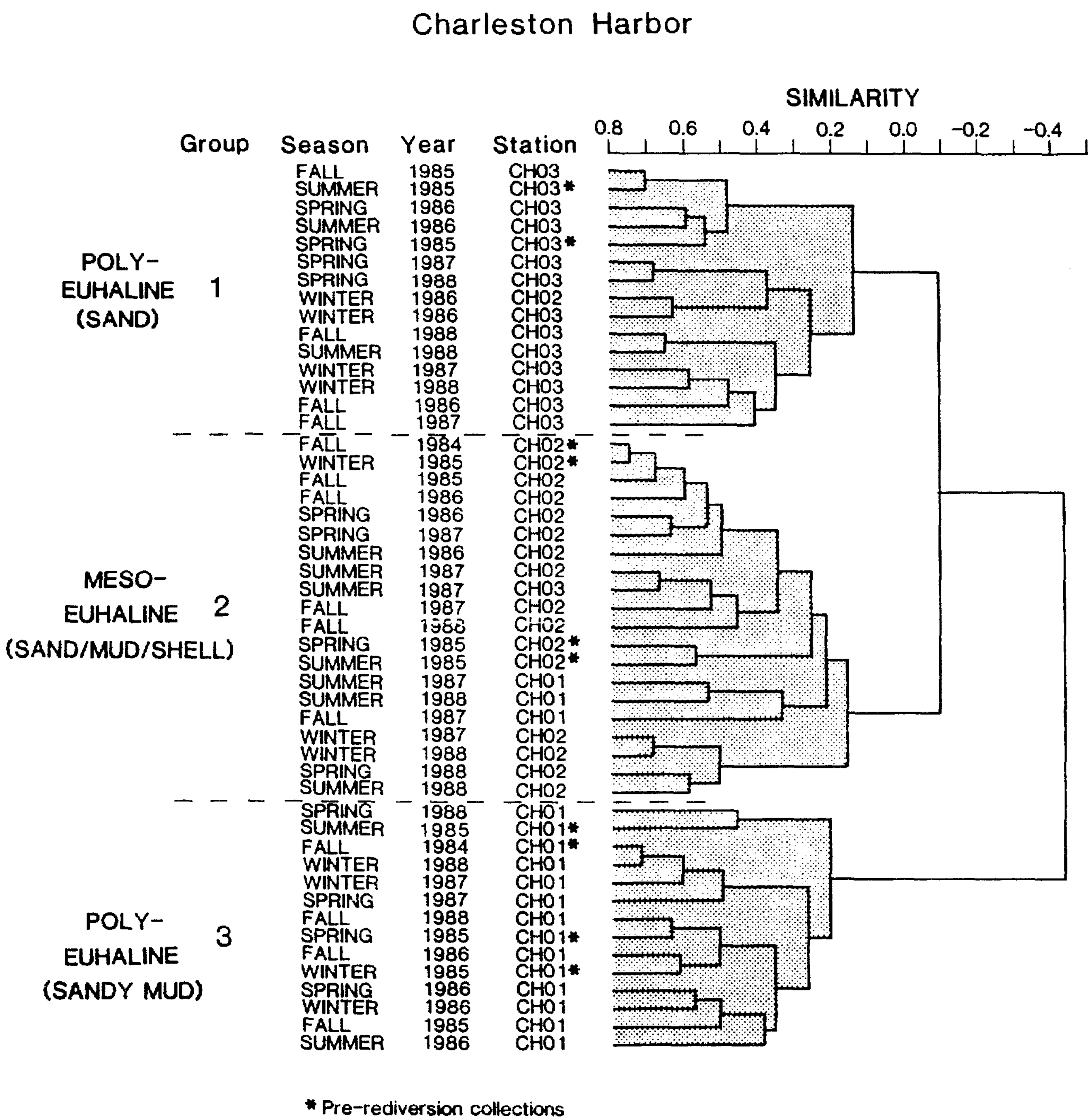


Figure VII.15. Hierarchical classification of Charleston Harbor grab samples generated by a normal cluster analysis.

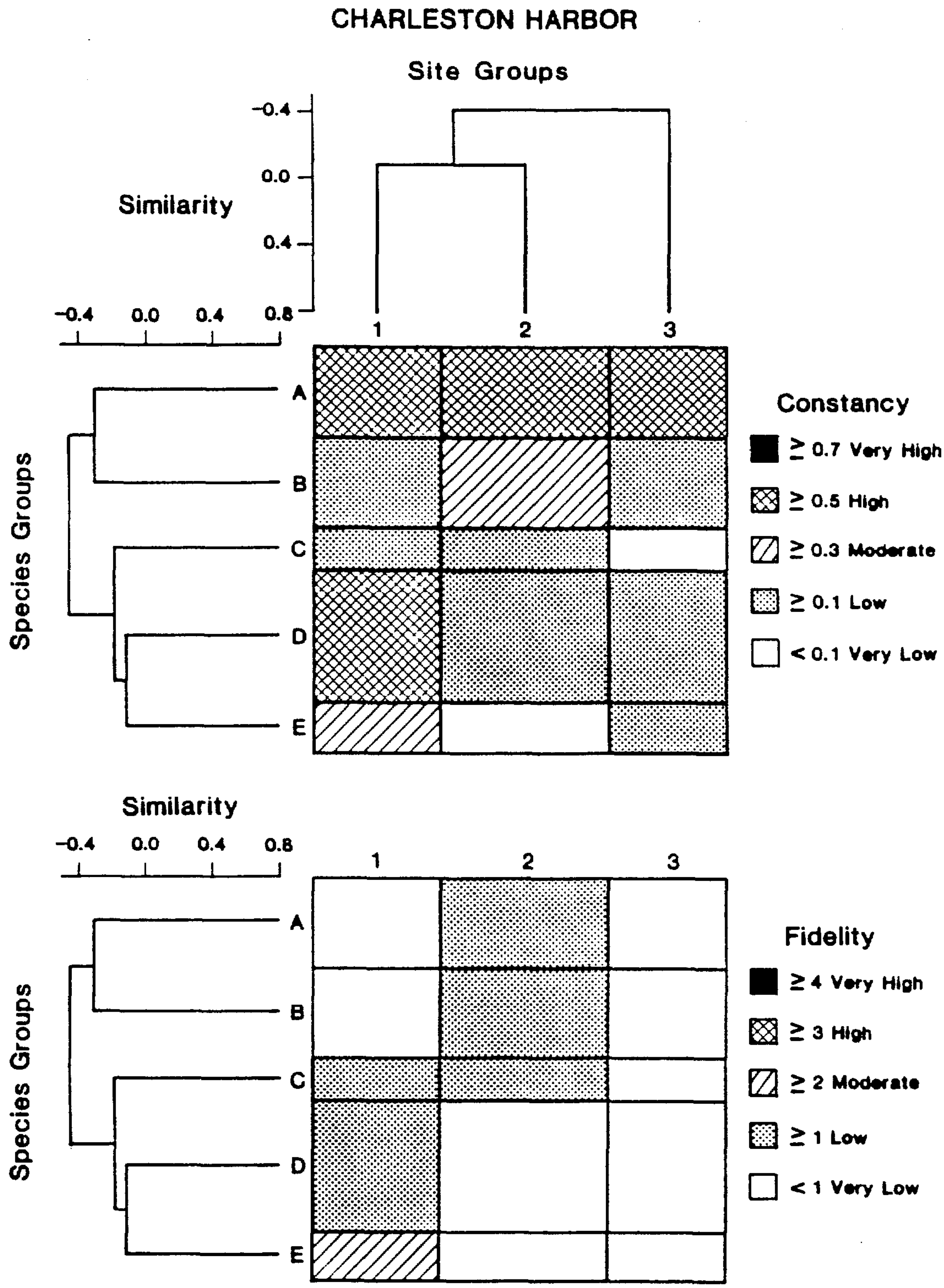


Figure VII.16. Nodal constancy and fidelity diagrams illustrating coincidences between species groups and site groups generated by inverse and normal cluster analyses of Charleston Harbor grab data.

Table VII.7. Species groups generated by an inverse cluster analysis of Charleston Harbor grab samples. (Am = amphipod; As = ascidian; B = bivalve; Cu = cumacean; D = decapod; G = gastropod; Is = isopod; My = mysid; N = nemertinean; Oc = octocoral; P = polychaete).

GROUP A

Paraprionospio pinnata (P)
Oligochaeta
Leucon americanus (Cu)
Heteromastus filiformis (P)
Nemertinea
Magelona phyllisae (P)
Glycinde solitaria (P)
Mulinia lateralis (B)
Mediomastus californiensis (P)
Edotea montosa (Is)
Petricola pholadiformis (B)

GROUP B

Ancistrosyllis jonesi (P)
Bhawania heteroseta (P)
Diopatra cuprea (P)
Paracaprella tenuis (Am)
Sabellaria vulgaris (P)
Nereis succinea (P)
Batea catharinensis (Am)
Sigambra tentaculata (P)
Ogyrides alphaerostris (D)
Ilyanassa obsoleta (G)
Molgula manhattensis (As)

GROUP C

Phoronida
Ostracoda
Cistenides gouldii (P)
Ampharete americana (P)
Turbonilla sp. G (G)

GROUP D

Glycera americana (P)
Carinomella lactea (N)
Scoloplos rubra (P)
Streblospio benedicti (P)
Tellina texana (B)
Nematoda
Listriella clymenellae (Am)
Clymenella torquata (P)
Acteocina canaliculata (G)
Abra aequalis (B)
Monoculodes sp. A (Am)
Eteone heteropoda (P)
Spiophanes bombyx (P)
Oxyurostylis smithi (Cu)
Aligena elevata (B)
Neomysis americana (My)

GROUP E

Renilla reniformis (Oc)
Cerapus tubularis (Am)
Pinnixa chaetopterana (D)
Listriella barnardi (Am)
Nucula proxima (B)
Notomastus latericeus (P)

often in sand and muddy sand habitats, perhaps accounting for their relatively high constancy among collections from the sandiest of the three sites, station CH03 (site group 1). The more psammophilic species in this group include the bivalve *Tellina texana*, the amphipod *Monoculodes* sp. A, the mysid *Neomysis americana*, and the polychaetes *Glycera americana*, *Spiophanes bombyx* and *Clymenella torquata* (Abbott, 1974; Bousfield, 1973; Gilbert, 1984; Maurer *et al.*, 1976; Wolff, 1973). While primary diversity was enhanced at station CH03 by the presence of sandy sediments, secondary diversity was enhanced by the presence of the tube-dwelling polychaete, *Clymenella torquata*, which served as the host for at least two commensal species, the amphipod *Listriella clymenellae* and the bivalve *Aligena elevata* (Bousfield, 1973; Fox and Ruppert, 1985). Although many of the species belonging to group D seem to prefer sandier substrates, others are fairly indiscriminate with respect to sediment type. These include the gastropod *Aceteocina canaliculata* and the polychaetes *Streblospio benedicti*, *Scoloplos rubra* and *Eteone heteropoda* (Pettibone, 1963; Roberts *et al.*, 1975; Taylor, 1984). The occurrence of these species in a variety of habitats (represented in this study by the three harbor basin sites) probably accounts for the low fidelity of this group among collections from any one site group.

Species group E was moderately constant among collections from station CH03 (site group 1). Like group D, species in group E are characterized by Fox and Ruppert (1985) as perennial inhabitants of protected beaches, creeks, and sounds. Two members of this group (the bivalve *Nucula proxima* and the capitellid polychaete *Notomastus latericeus*) are eurytopic with respect to sediment type; however, two other species (the sea pansy *Renilla reniformis* and the tubicolous amphipod *Cerapus tubularis*) are generally restricted to sand and muddy sand habitats (Bousfield, 1973; Fox and Ruppert, 1985). These sediment preferences may explain the moderate fidelity of this species group to collections from the sandiest of the three harbor basin sites, station CH03. Finally, two other members of species group E (the pinnotherid crab *Pinnixa chaetopterana* and the amphipod *Listriella barnardi*) are known to be commensal inhabitants of tubes constructed by terebellids, chaetopterids, and other marine polychaetes (Bousfield, 1973; Frankenberg and Leiper, 1977). Since the distributions of these species did not appear to coincide with those of any tubicolous polychaetes, it is uncertain whether or not these species were indeed living commensally with other organisms.

As in the Cooper and Wando Rivers, there were no consistent patterns of similarity among collections with respect to season or year. Nor were there any differences in overall community structure that could be related to rediversion. A few of the numerically dominant species did, however, exhibit changes in relative abundance that seemed to coincide with rediversion, or occur shortly thereafter (Figures VII.17a through VII.17j). As in the Cooper and Wando Rivers, the polychaete *Paraprionospio pinnata* became much

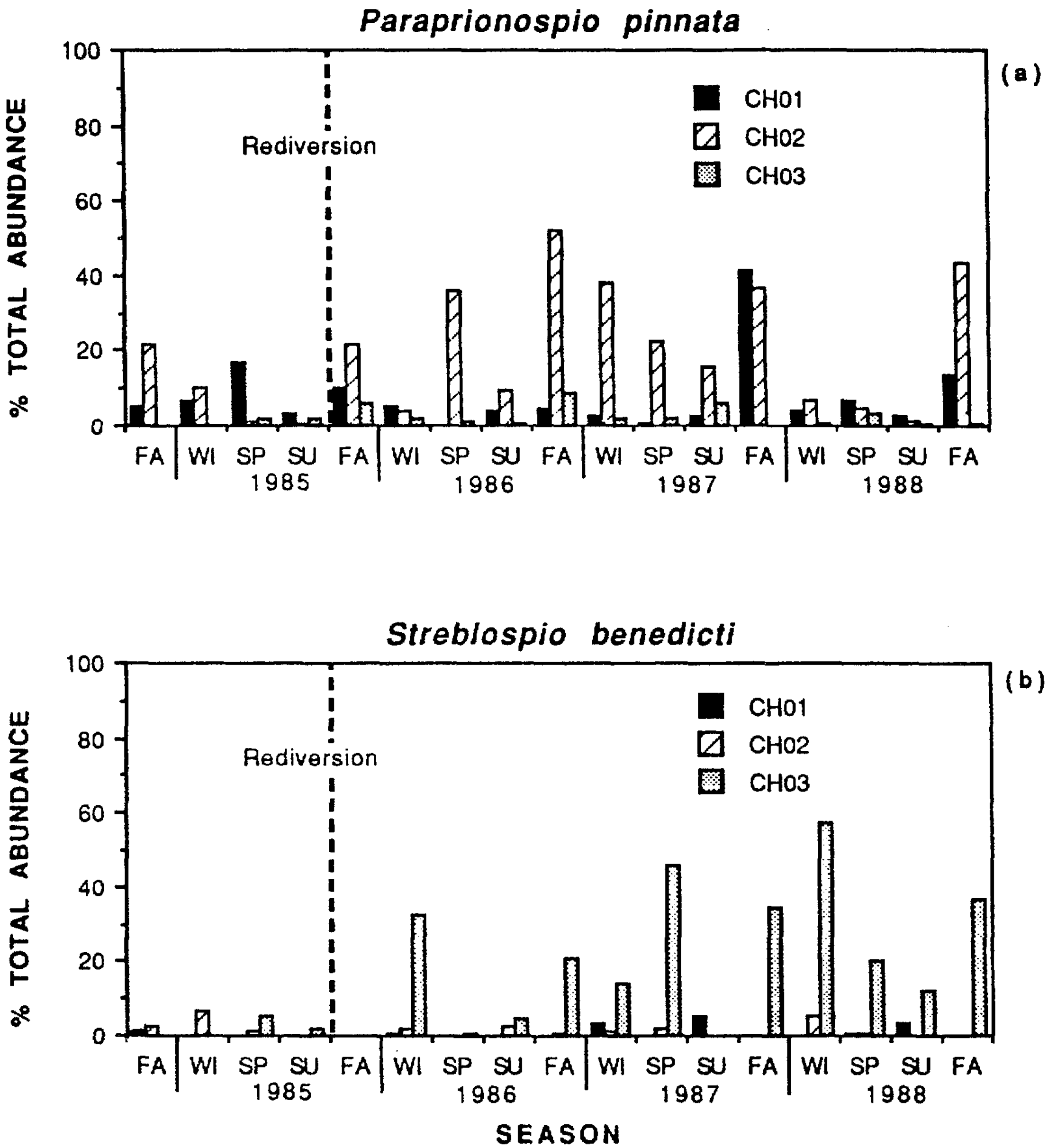


Figure VII.17. Percentage of total faunal abundance contributed by each of the 10 most abundant species in pooled replicate grab samples collected from each of the Charleston Harbor sites during each season of the four-year study.

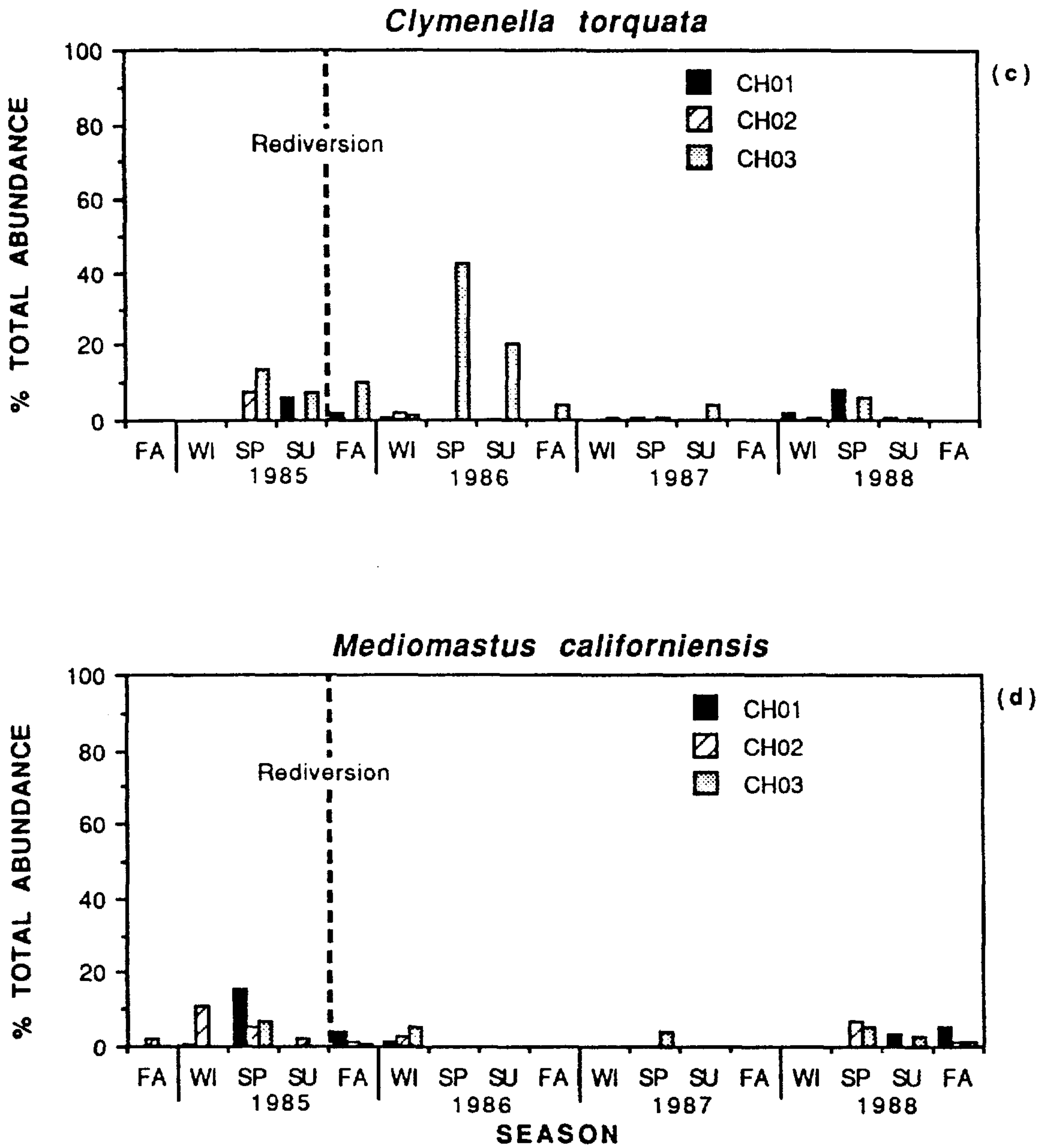
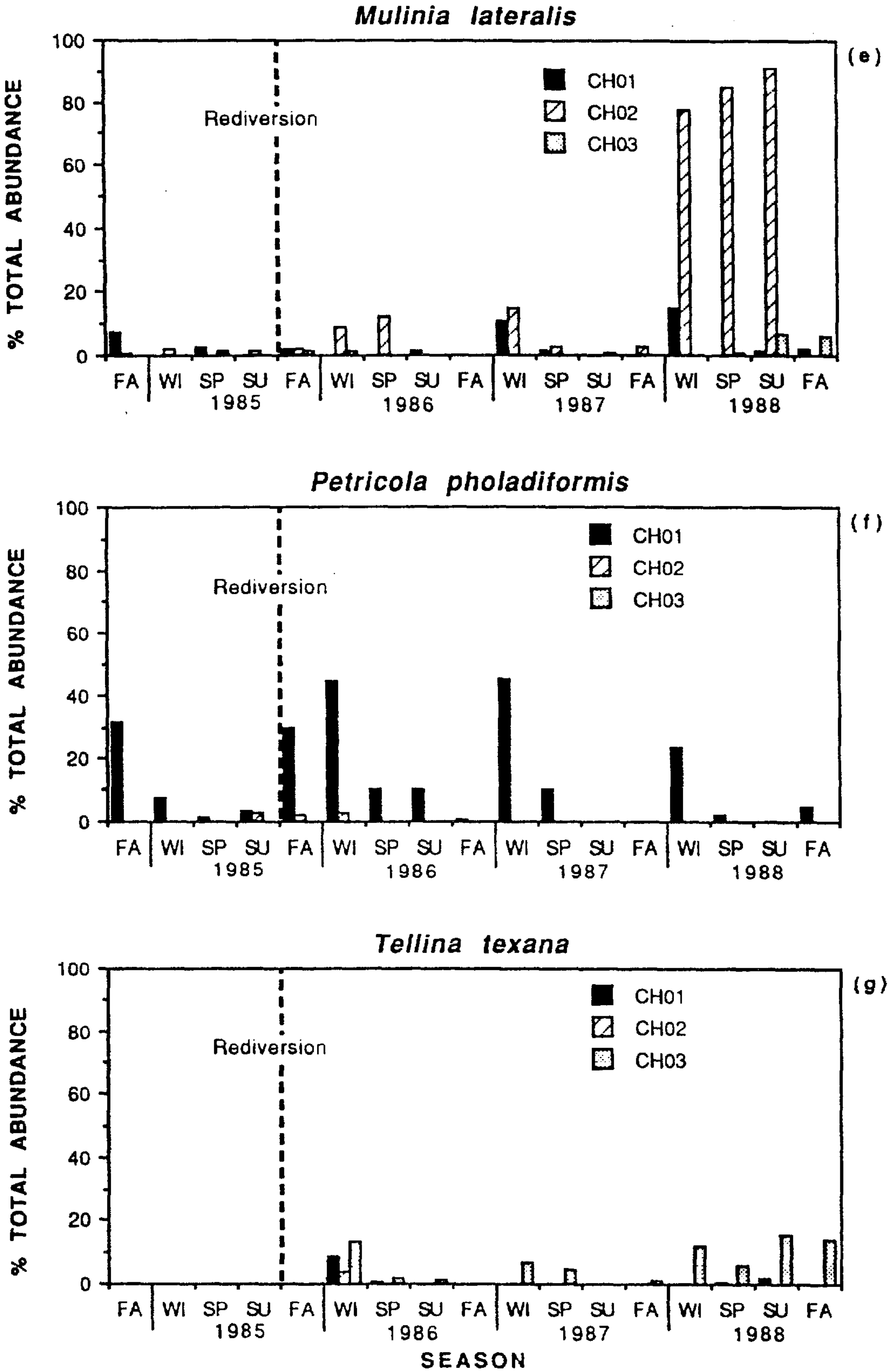


Figure VII.17. (Continued)



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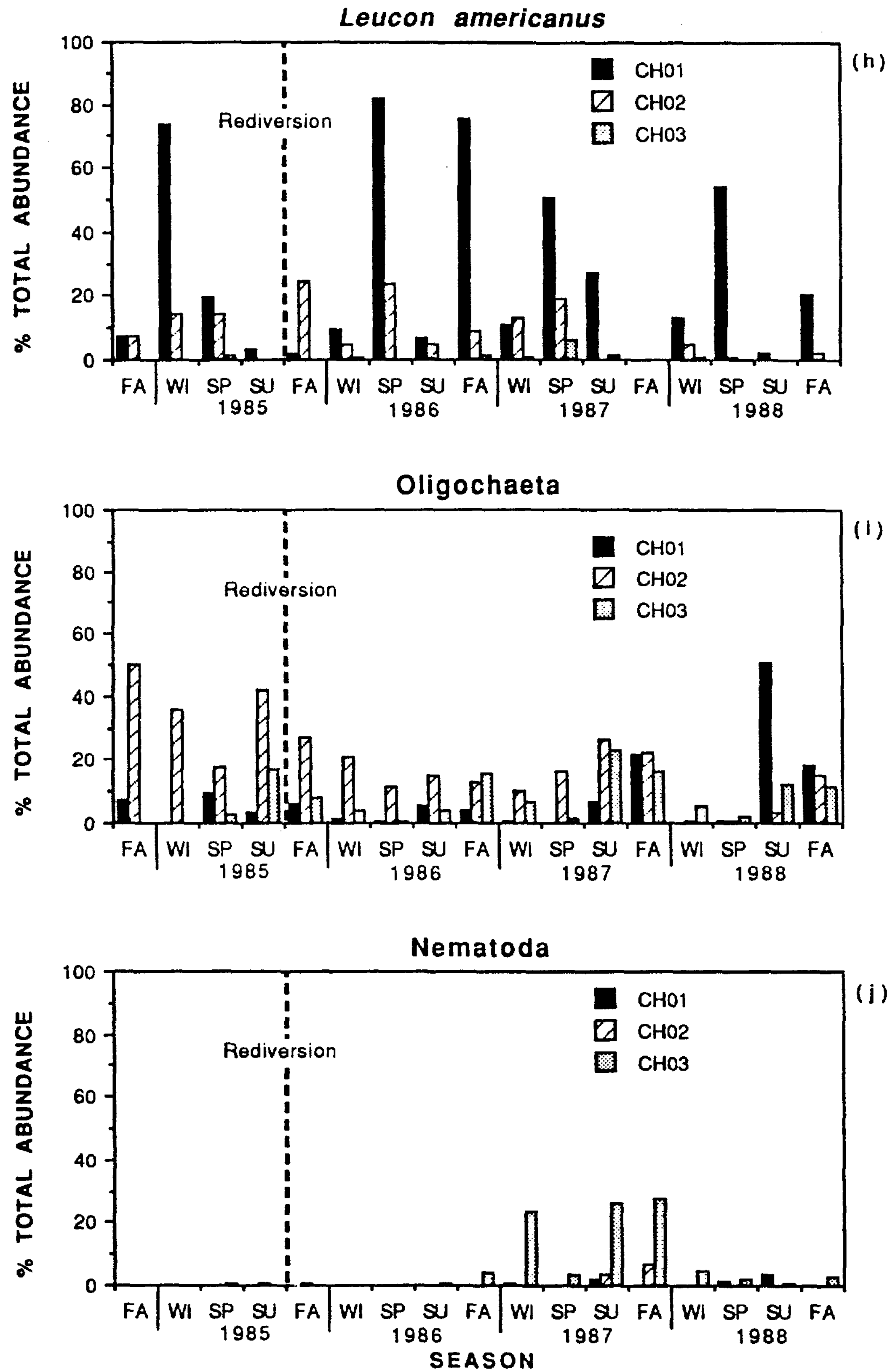


Figure VII.17. (Continued)

more abundant at one of the harbor sites (station CH02) subsequent to rediversion (Figure VII.17a). Possible explanations for the increase in abundance of this species were discussed in the previous two sections. The subsequent decline in relative abundance of *P. pinnata* at station CH02 during the first three seasons of 1988 reflects an irruptive increase in numbers of the opportunistic bivalve *Mulinia lateralis*, rather than a true decrease in the actual abundance of *P. pinnata* (Figure VII.17e).

The specific cause of the dramatic rise and fall in numbers of *M. lateralis* is unknown; however, it is not an unusual occurrence (Boesch *et al.*, 1976; Grassle and Grassle, 1974; Walker and Tenore, 1984; Wright *et al.*, 1983). Its short generation period, high reproductive rate and planktonic dispersal of larvae enable *M. lateralis* to rapidly colonize new substrates in great numbers (Grassle and Grassle, 1974; Calabrese, 1969). Although *M. lateralis* is considered to be euryhaline, both with respect to its geographical distribution and its physiological tolerance as demonstrated in the laboratory (Williams, 1984), the larval development, survival and growth of this species are optimal within a relatively narrow range of salinities (Walker and Tenore, 1984). It may be that these conditions prevailed at station CH02 during the first three quarters of 1988 when *M. lateralis* was found so abundantly. The drastic decline in abundance of *M. lateralis* in other estuaries has been attributed to mass mortalities resulting from several different factors, including 1) clogging of the clam's filtering apparatus by large volumes of particulate matter forced into suspension by the increased activities of deposit feeders in summer (Boesch *et al.*, 1976; Wright *et al.*, 1983); 2) periodically low dissolved oxygen concentrations (Holland *et al.*, 1989); and 3) seasonal declines in food resources (Shumway and Newell, 1984).

While the polychaete *Paraprionospio pinnata* increased in relative abundance at station CH02 subsequent to rediversion, oligochaetes decreased (Figure VII.17i). This decrease was, in part, a reflection of the increased numbers of *P. pinnata*; however, oligochaetes decreased in absolute numbers as well.

Ashley River:

Unlike the other two rivers or the harbor basin, the Ashley River was sampled for macrofauna only during the last year of our four-year study. Consequently, no conclusions can be drawn regarding the effects of rediversion on this river. However, in the absence of any yearly variation in the data (with the exception of some discrepancies between Fall, 1987 and Fall, 1988 samples), seasonal and site-related patterns of abundance and distribution become more apparent.

The 45 grab samples taken at the three Ashley River sites from Fall, 1987 through Fall, 1988 contained a total of 3,129 organisms representing 78 species. Mean abundances ranged from a low of 9.33 individuals/grab at station AR02 in Fall, 1987 to a high of 245.67/grab at station AR03 in Spring, 1988. Abundances were usually highest at station AR03, the site furthest upriver, and were generally lowest at station AR02 (Figure VII.18). Mean abundances at stations AR01 and AR03 were highest in spring, but at station AR02 they were highest in summer.

The results of a two-way analysis of variance confirmed that the total abundance of organisms in the Ashley River differed significantly, both among sites and among seasons (Table VII.8). *A posteriori* comparisons of main effect group means indicated that abundances were significantly lower at station AR02 than at either of the other two sites and were significantly higher in spring than in any other season (Table VII.9).

Species diversity (H') values ranged from a low of 1.57 bits/individual at station AR02 in Fall, 1988 to a high of 2.97 bits/individual at station AR03 in Winter, 1988. These extremes encompassed a much narrower range of diversity values than that encountered in the other two rivers or the harbor basin. There were no consistent patterns of species diversity (H') or species evenness (J') values with respect to site or season; however, species richness values (S-1/ln N) were, with one exception, higher at station AR01 than at either of the other two sites (Figure VII.19). Nevertheless, samples from stations AR02 and AR03 frequently had higher indices of diversity than those from AR01 due to the more equitable distribution of individuals among those species present at the former two sites.

A two-way analysis of variance comparing the total number of unique species among sites and seasons showed season to be significant factor, but not site (Table VII.10). *A posteriori* comparisons of group means showed the total number of species to be significantly greater in spring than in winter or fall, but not summer (Table VII.11). Furthermore, although the total number of species was greater at station AR01 than at either of the other two sites, this difference was not statistically significant ($p = 0.05$).

The results of a normal cluster analyses of pooled replicate grab samples from the Ashley River generated four site groups (Figure VII.20). Samples from station AR03 (site group 1) clustered separately from those taken at stations AR01 and AR02 (site groups 2, 3, and 4). The dissimilarity between samples from AR03 and those from the other two sites was largely a function of the lower salinity and sandier sediments at the upriver site (Figures III.14 and VII.6). Nevertheless, sediment type appeared to be of secondary importance in determining benthic community structure, judging from the close similarity in species composition between samples from stations AR01 and AR02. Sediments at

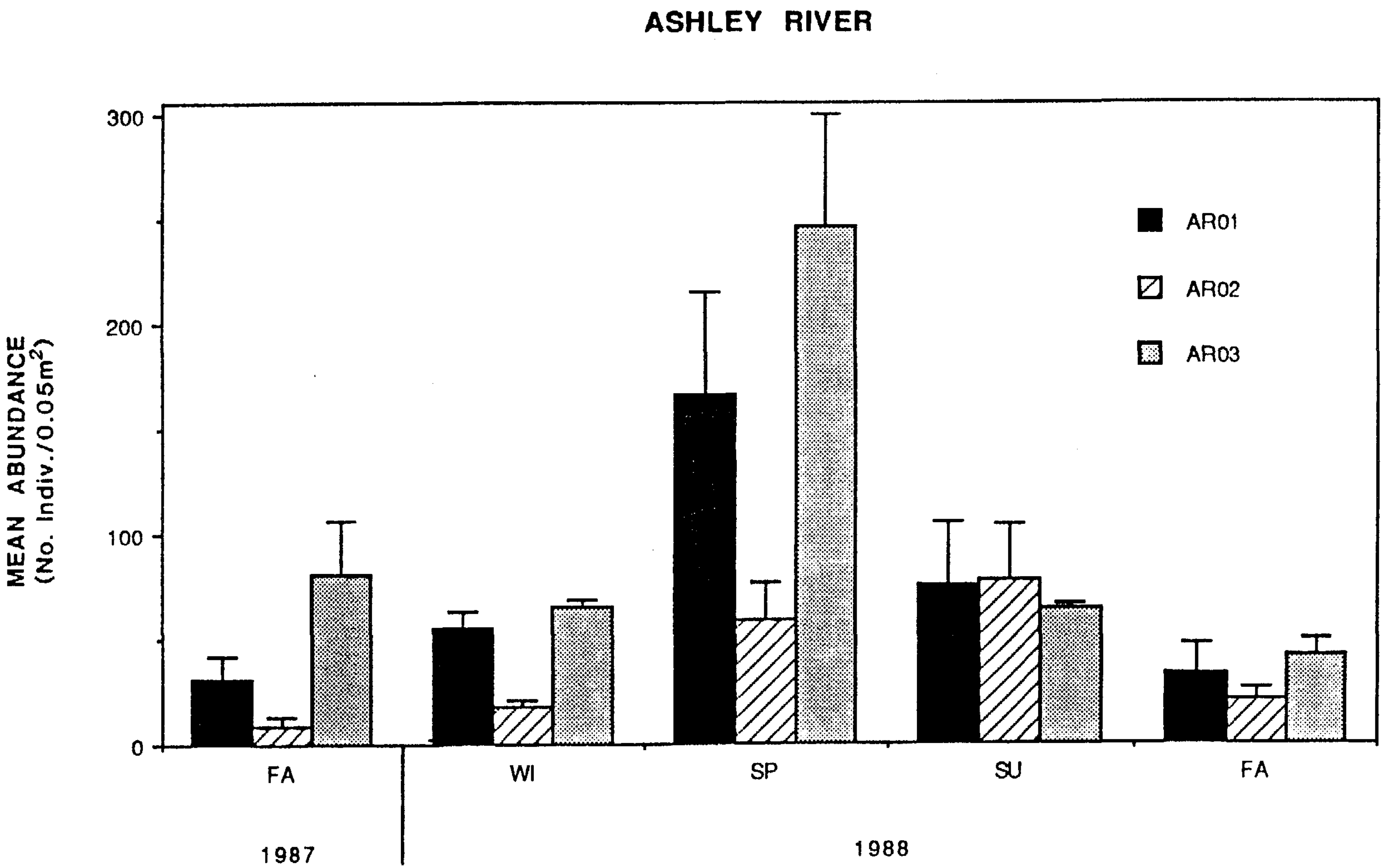


Figure VII.18. Mean abundance of macrofaunal organisms collected at each of the Ashley River sampling sites during each season of the last year of the four-year study. (Vertical lines represent one standard error of the mean.)

Table VII.8. Results of a two-way (Model I) analysis of variance comparing the mean numbers of macrofaunal organisms per grab sample among sites and seasons within the Ashley River.

Dependent Variable: Log_{10} (total no. indiv. +1)

	Ashley River	
Source of Variation	df	F_s
Site	2	14.62 ***
Season	3	13.47 ***
Site x Season	6	2.26 N.S.

*** p≤0.001

Table VII.9. Results of *a posteriori* comparisons (Student-Newman-Keuls Tests) of main effect group means+ for the Ashley River.

Dependent Variable: Log_{10} (total no. indiv. +1)

Source of Variation				
Site (n = 12)	26.28 μAR02	61.13 μAR01	90.57 μAR03	
Season (n = 9)	24.50 μFA	38.63 μWI	64.74 μSU	124.52 μSP

+ Group means are re-transformed to the linear scale; those means connected by underlines are not significantly different at p = 0.05.

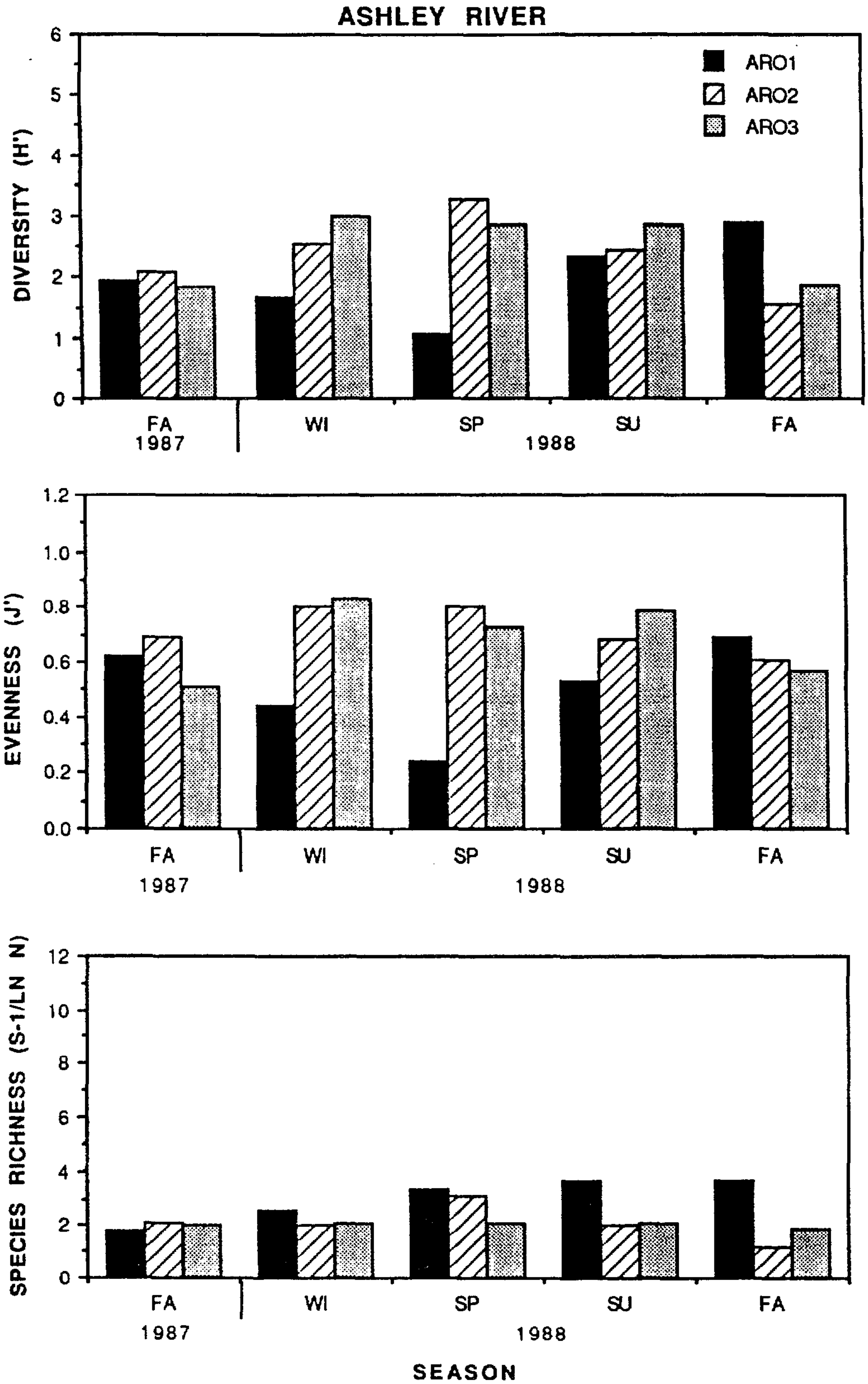


Figure VII.19. Species diversity (H'), evenness (J') and richness (S-1,ln N) values for benthic macrofaunal samples collected at each of the Ashley River stations during each season of the last year of the four-year study.

Table VII.10. Results of a two-way (Model I) analysis of variance (without replication) comparing the total number of unique species in each set of replicate samples among sites and seasons within the Ashley River.

Dependent Variable: Log_{10} (total no. species +1)

Source of Variation	Ashley River df	F_s
Site	2	2.96 N.S.
Season	3	5.48 *
Site x Season	6	-

* p≤0.05

Table VII.11. Results of *a posteriori* comparisons (Student-Newman-Keuls Tests) of main effect group means+ for the Ashley River.

Dependent Variable: Log_{10} (total no. species +1)

Source of Variation				
Site (n = 4)	11.56 $^{\mu}$AR02	13.45 $^{\mu}$AR03	16.42 $^{\mu}$AR01	
Season (n = 3)	9.47 $^{\mu}$FA	12.80 $^{\mu}$WI	15.60 $^{\mu}$SU	18.05 $^{\mu}$SP

+ Group means are re-transformed to the linear scale; those means connected by underlines are not significantly different at p = 0.05.

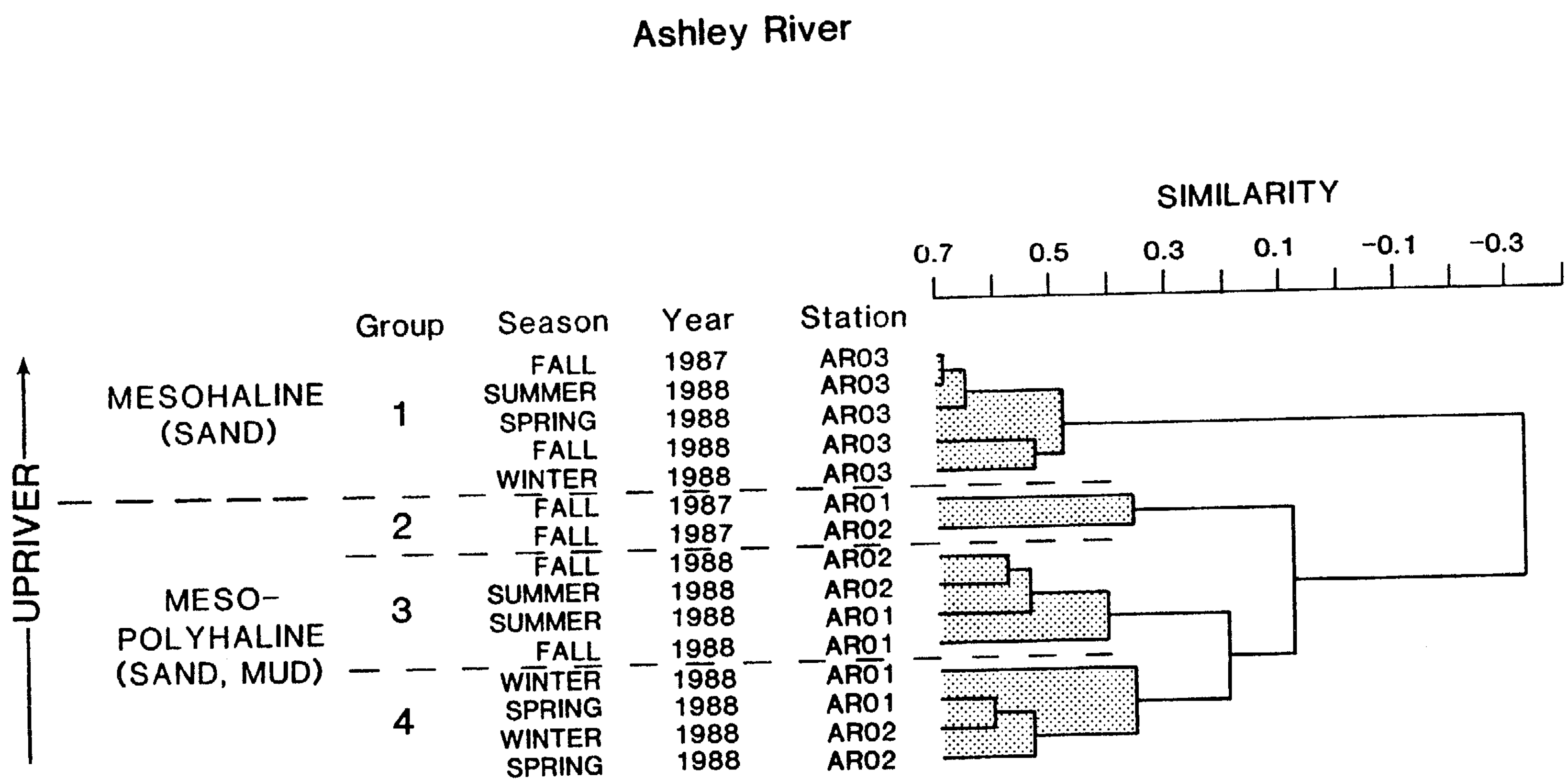


Figure VII.20. Hierarchical classification of Ashley River grab samples generated by a normal cluster analysis.

AR02 (like those at AR03) were predominantly sand; whereas, sediments at AR01 were predominantly silt/clay. Although there were no clear seasonal patterns among samples from station AR03, summer and fall collections taken at AR01 and AR02 (site group 3) clustered separately from winter and spring collections taken at the same two sites (site group 4). Furthermore, Fall 1987 collections (site group 2) clustered separately from Fall, 1988 collections taken at AR01 and AR02.

Results of the inverse cluster and nodal analyses show which species accounted for these seasonal and site-related differences among grab samples (Figure VII.21; Table VII.12). Species group A included several species that were very highly constant among, but not particularly faithful to, samples from station AR03 (site group 1). Some of the species in this group are euryhaline inhabitants of sandy substrates throughout the southeast (Bousfield, 1973; Croker, 1967; Roberts *et al.*, 1975). These include the isopod *Cyathura polita* as well as the amphipods *Lepidactylus dytiscus* and *Monoculodes* sp. A. At least one species in this group, the polychaete *Heteromastus filiformis*, is eurytopic with respect to both salinity and sediment type; however, in mesohaline salinities such as those that prevail at station AR03, it is reportedly most common on sand bottoms (Roberts *et al.*, 1975). Unlike the foregoing members of species group A, the spionid polychaete *Scolecolepides viridis* is more restricted in its distribution with respect to salinity, rarely occurring in the high mesohaline to polyhaline salinities characteristic of stations AR01 and AR02 (Roberts *et al.*, 1975).

Species group B included two estuarine endemic amphipods, *Gammarus tigrinus* and *Corophium lacustre*, as well as an unidentified turbellarian. Both amphipods are common inhabitants of the lower salinity portions of estuaries, thus accounting for their moderate constancy and fidelity among samples from upriver station AR03 (site group 1).

Species group C included two capitellid polychaetes, *Mediomastus californiensis* and *Capitella capitata*. This group was highly faithful to, and consistently occurred in, samples collected from stations AR01 and AR02 in the summer and fall of 1988 (site group 3). While both species are more commonly found in fine and muddy sand, they also occur in medium sand and mud (Ewing, 1984; Wolff, 1973). Their relative indifference to substrate may explain their ability to occupy the wide range of sediments encompassed by collections from the two downriver sites.

Although both of these polychaetes are perennial inhabitants of South Carolina waters (Fox and Ruppert, 1985), *C. capitata* is found relatively infrequently. This species, which occurs abundantly in areas of high salinity where large amounts of organic material have been deposited, has been cited as an indicator of pollution or environmental

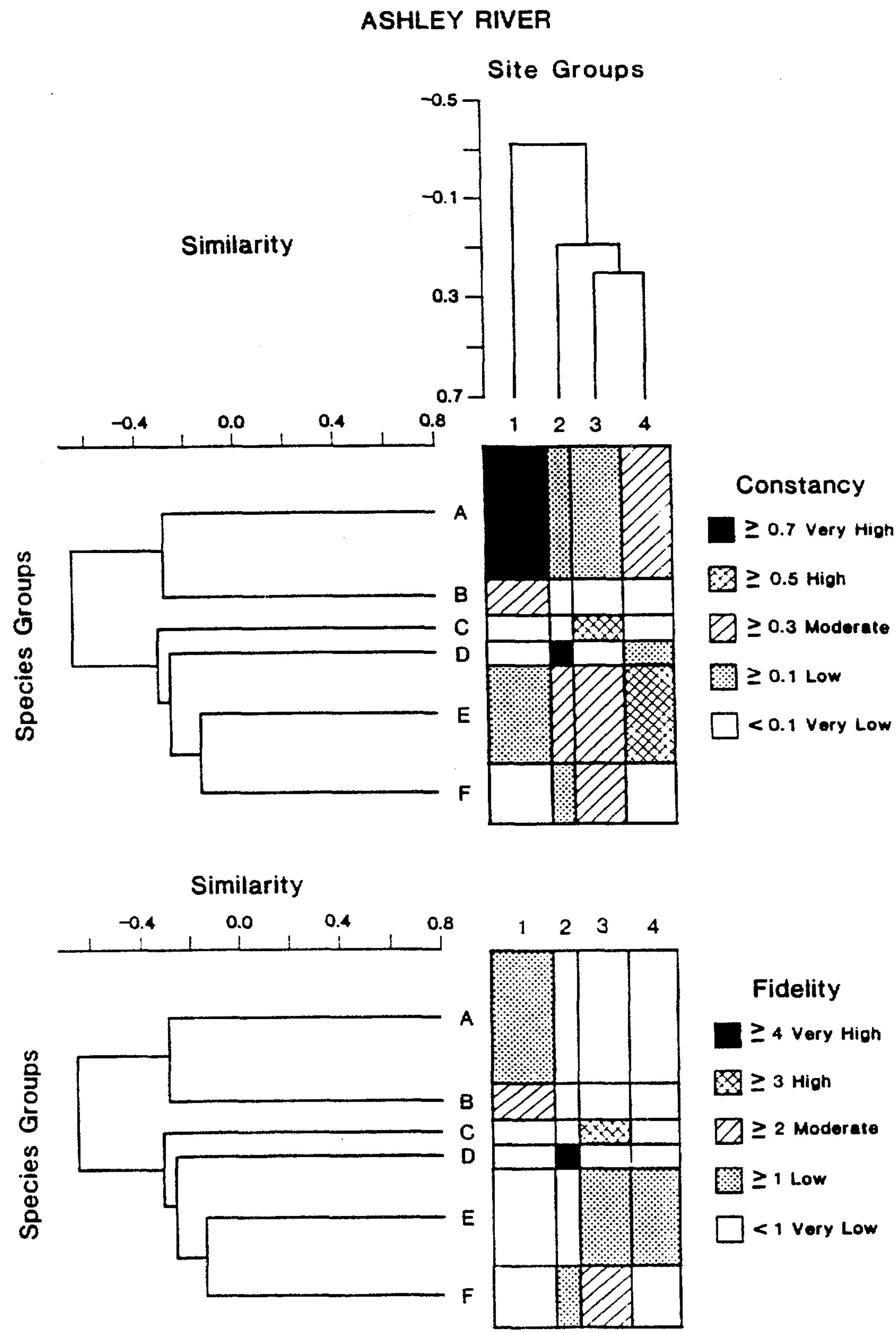


Figure VII.21. Nodal constancy and fidelity diagrams illustrating coincidences between species groups and site groups generated by inverse and normal cluster analyses of Ashley River grab data.

Table VII.12. Species groups generated by an inverse cluster analysis of Ashley River grab samples. (Am = amphipod; B = bivalve; Cu = cumacean; C = decapod; G = gastropod; Is = isopod; P = polychaete).

GROUP A

Lepidactylus dytiscus (Am)
Chiridotea almyra (Is)
Scolecolepides viridis (P)
Cyathura polita (Is)
Heteromastus filiformis (P)
Oligochaeta
Nemertinea
Nematoda
Monoculodes sp. A (Am)
Cyclaspis varians (Cu)

GROUP B

Gammarus tigrinus (Am)
Turbellaria B
Corophium lacustre (Am)

GROUP C

Mediomastus californiensis (P)
Capitella capitata (P)

GROUP D

Ogyrides alphaerostris (D)
Mancocuma sp. (Cu)

GROUP E

Leucon americanus (Cu)
Ostracoda
Edotea montosa (Is)
Nereis succinea (P)
Oxyurostylis smithi (Cu)
Mulinia lateralis (B)
Streblospio benedicti (P)
Paraprionospio pinnata (P)

GROUP F

Ilyanassa obsoleta (G)
Ampelisca abdita (Am)
Phoronida
Spiochaetopterus oculatus (P)
Sabellaria vulgaris (P)

disturbance (Wolff, 1973; Grassle and Grassle 1974). Its apparent resistance to low dissolved oxygen levels would be particularly adaptive in this type of environment (Wolff, 1973), and may explain its occurrence at station AR02 during the summer of 1988 when dissolved oxygen levels were relatively depressed (Figure III.19). Although *C. capitata* never occurred in great abundance in the Ashley River, the fact that this relatively rare species was present at all suggests that this river may be periodically stressed by high inputs of organic matter and low dissolved oxygen levels.

Species group D included two species (the decapod *Ogyrides alphaerostris* and the cumacean *Mancocuma* sp.) that were very highly faithful to collections from stations AR01 and AR02 taken during the fall of 1987 (site group 2). Although nothing is definitively known about the habitat requirements of *Mancocuma* sp., cumaceans, in general, tend to inhabit a narrow range of sediments (Watling, 1979). *Ogyrides alphaerostris* is similarly restricted by sediment type, being most often found in very fine sand or mud (Frankenberg and Leiper, 1977). While sediments at station AR02 were predominantly sandy during four of the five seasons sampled, they were substantially muddier (almost 50% silt/clay) in Fall, 1987 when these two species were most abundant (Figure VI.6).

Species group E exhibited high constancy, but low fidelity, among winter and spring collections from stations AR01 and AR02 (site group 4). This species group contains many of the most abundant and ubiquitous species found throughout the meso- and polyhaline zones of the Charleston Harbor estuary. Although most of these species are eurytopic with respect to sediment type, several (including the polychaetes *Paraprionospio pinnata* and *Nereis succinea*, the cumacean *Leucon americanus* and the bivalve *Mulinia lateralis*) seem to prefer the muddy sediments typical of station AR01 (Johnson, 1984; Roberts *et al.*, 1975; Thayer, 1975). Fox and Ruppert (1985) describe most of these species as perennial constituents of the benthos along the South Carolina coast. However, it appears from this study that they are more commonly found in winter and spring, at least in the Ashley River.

Finally, group F included several species that showed a high level of constancy among summer and fall collections from stations AR01 and AR02 (site group 3). In fact, most of these species were almost entirely restricted to station AR01, the site furthest downriver. Many of the species in group F (including two infaunal tube-dwellers, the amphipod *Amplisca abdita* and the polychaete *Spiochaetopterus oculatus*, as well as the mud snail *Ilyanassa obsoleta*) are commonly found in sediments ranging from fine or muddy sand to silt/clay in polyhaline to euhaline salinities (Bousfield, 1973; Fox and Ruppert, 1985; Gilbert, 1984a). The occurrence at AR01 of another tube-dwelling polychaete, *Sabellaria vulgaris*, may be evidence of the nearby oyster banks which probably serve as a source of hard substrate for the attachment of this, and perhaps other, epifaunal organisms. Most of the species in group F, like those in group E, have been described as year-around inhabitants of South Carolina coastal waters (Fox and Ruppert, 1985); however, *A. abdita* is reportedly most abundant in spring (Bousfield, 1973; Wendt *et al.*, in prep.). This pattern is consistent with its seasonal occurrence in the Ashley River.

Among the numerically dominant species, *P. pinnata* was most abundant at the middle- and downriver stations, AR02 and AR01, in the summer and fall of 1988 (Figure VII.22a). As in the Cooper and Wando Rivers, the sites at which *P. pinnata* was most abundant in the Ashley River differed greatly in sediment type but not salinity regime suggesting, once again, that this species is more eurytopic with respect to sediment grain size than it is with respect to salinity.

Another spionid polychaete, *Streblospio benedicti*, was most abundant in the Ashley River during the fall of 1987 when it accounted for approximately 60% of the total abundance of organisms at station AR01; however, this species was poorly represented in other seasons and was entirely absent from collections taken the following fall (Figure VII.22b). Conversely, the bivalve *Mulinia lateralis* was very abundant at stations AR01 and

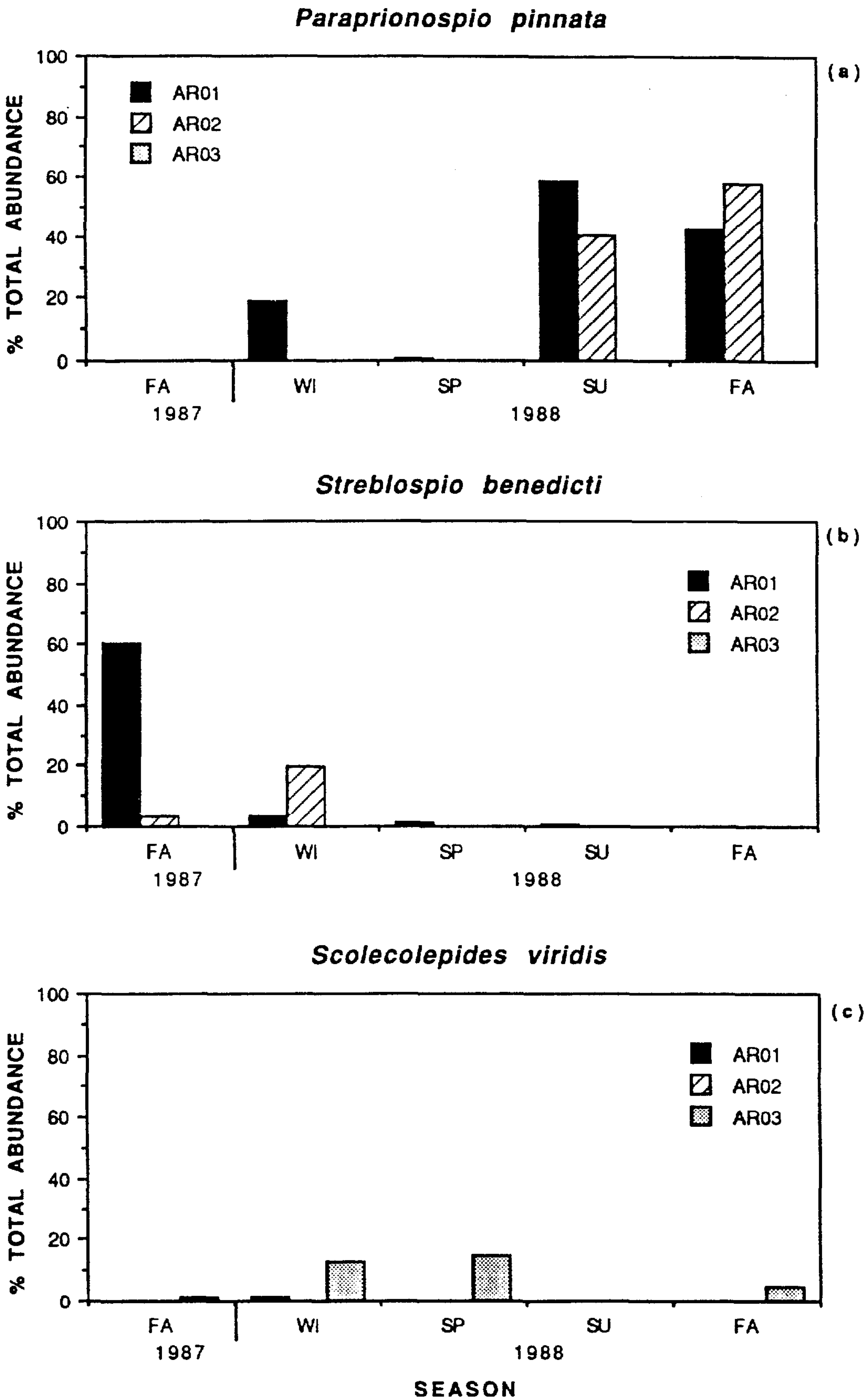
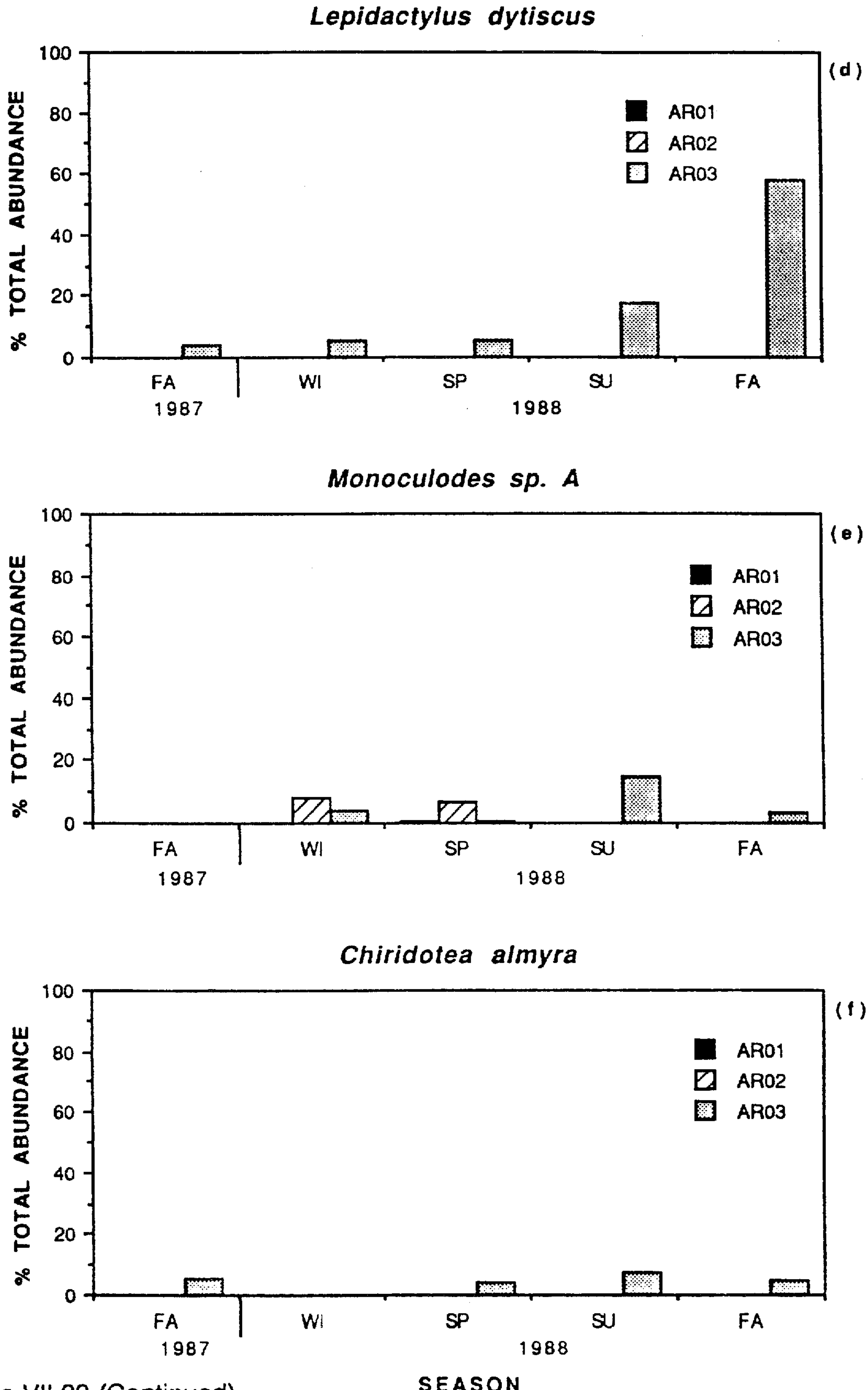


Figure VII.22. Percentage of total faunal abundance contributed by each of the 10 most abundant species in pooled replicate grab samples collected from each of the Ashley River sites during each season of the last year of the four-year study.



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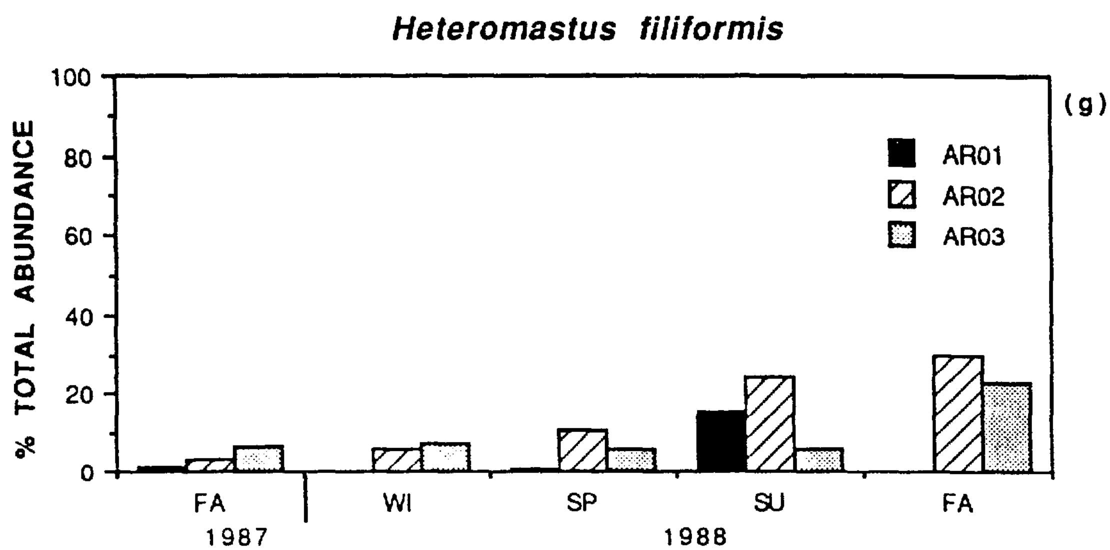


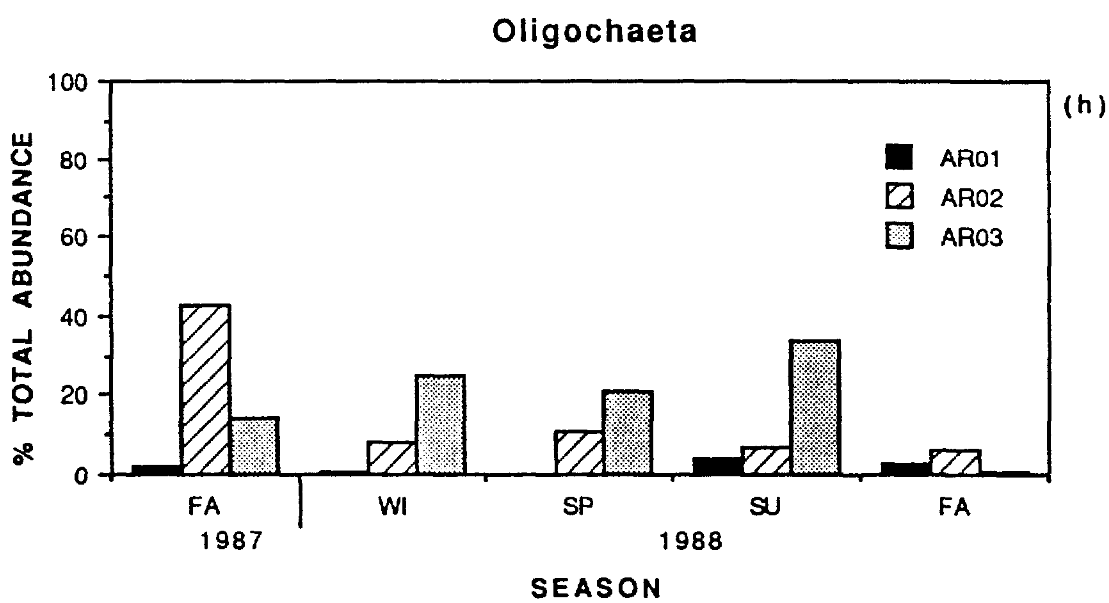
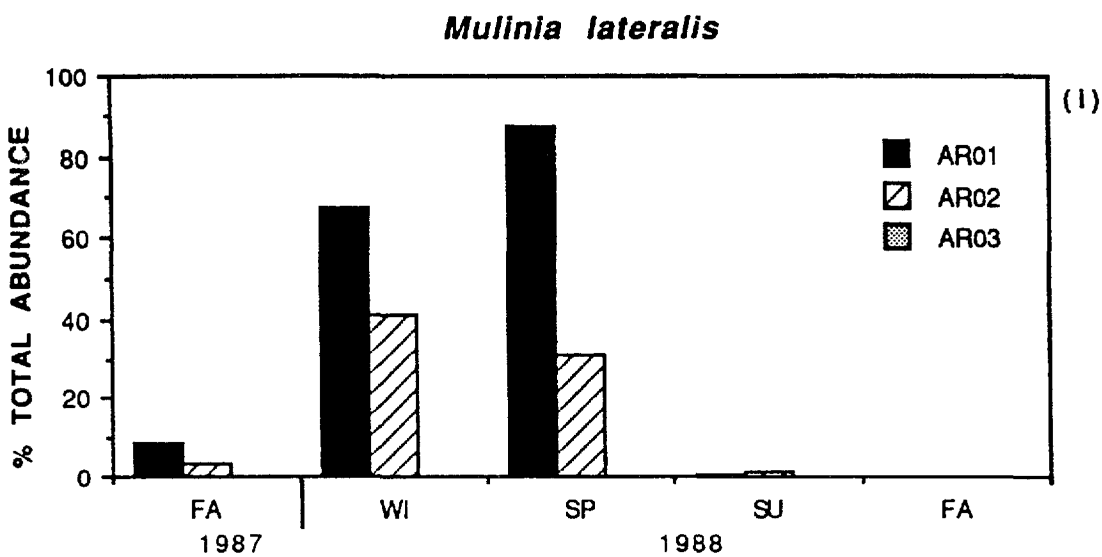


Figure VII.22 (Continued)



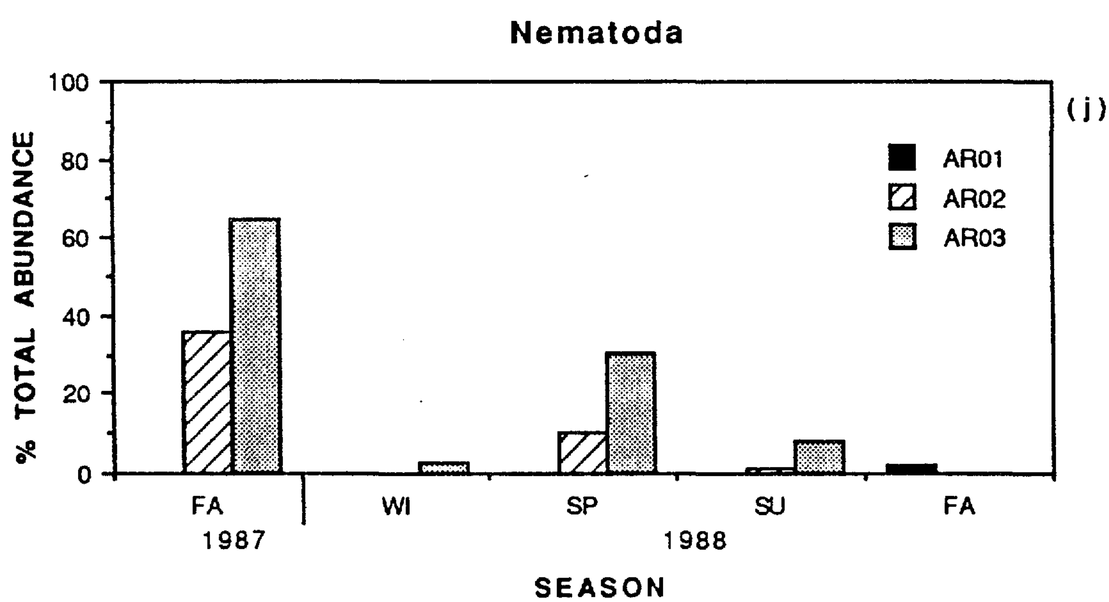


Figure VII.22. (Continued)

AR02 in winter and spring, but not in summer or fall (Figure VII.22i). Both of these opportunistic species were also sporadically abundant in the Cooper and Wando Rivers, as well as the harbor basin, but displayed no consistent seasonal trends in abundance anywhere.

Finally, several numerically dominant species were mostly restricted to sand bottoms in the middle and upper reaches of the Ashley River. These included the polychaete *Scolecolepides viridis*, the amphipods *Lepidactylus dytiscus* and *Monoculodes* sp. A and the isopod *Chiridotea almyra* (Figures VII.22c through VII.22f). Oligochaetes and the capitellid polychaete *Heteromastus filiformis* occurred somewhat more frequently than the foregoing species at downriver site AR01 but, like those species, were generally most abundant at the sites further upriver, stations AR02 and AR03 (Figures VII.22g and VII.22h). These patterns of distribution and abundance are consistent with the habitat preferences discussed earlier.

Synopsis of Long-Term Spatial and Temporal Patterns:

The results of our four-year study indicate that the spatial distribution of benthic macrofauna in the Charleston Harbor estuary is similar, in many respects, to that of other gradient estuaries along the mid-Atlantic, southeast and Gulf coasts of the United States (Boesch, 1977b; Calder *et al.*, 1977; Dorges, 1977; Flint and Kalke, 1985; Holland *et al.*, 1987, 1989). In our study, as well as in these earlier investigations, salinity appears to be the most important determinant of benthic community structure, with sediment type playing a secondary, more localized role.

This was particularly evident in the Cooper River where there was a clear distinction among tidal freshwater, low salinity (oligo-mesohaline) and high salinity (meso-polyhaline) brackish water assemblages. Although salinities did not span as broad a range in the Ashley River, there was a similar discontinuity between low salinity and high salinity brackish water faunal groups. Salinities in the Wando River and harbor basin were generally in the high meso- to euhaline range and exhibited even less spatial variability than in the Ashley River. Accordingly, sediment type and biotic interactions appeared to be of greater importance in determining differences in the species composition and abundance of benthic organisms in these reaches of the estuary.

Limnetic portions of the Charleston Harbor estuary were dominated by insect larvae and oligochaetes. Low salinity (oligo- mesohaline) reaches were dominated by psammophilic species (*e.g.*, the amphipod *Lepidactylus dytiscus*) and estuarine endemics (*e.g.*, the isopods *Cyathura polita* and *Chiridotea almyra*; the amphipods *Gammarus tigrinus* and *Corophium lacustre*; and the polychaete *Scolecolepides viridis*). Higher salinity (meso-polyhaline) portions of the estuary were dominated by euryhaline marine and euryhaline opportunistic species (*e.g.*, the bivalve *Mulinia lateralis* and the polychaetes *Paraprionospio pinnata, Heteromastus filiformis, Mediomastus californiensis, Nereis succinea, Glycinde solitaria* and *Streblospio benedicti*). Stenohaline marine species (*e.g.*, the polychaetes *Clymenella torquata, Spiochaetopterus oculatus, Diopatra cuprea* and *Notomastus latericeus*) were generally restricted to the harbor basin and lower reaches of the Ashley, Cooper and Wando Rivers.

Many of the forenamed species, particularly those in the upper and middle reaches of the estuary, are identical to those inhabiting similar salinity regimes in estuaries throughout the middle Atlantic and southeastern United States (Boesch, 1977b; Calder *et al.*, 1977; Holland *et al.*, 1987, 1989). This is testimony to the cosmopolitan distribution of many euryhaline and estuarine endemic species.

Within each salinity zone, benthic assemblages were further distinguished by their affinities for different sediment types. Differences in species composition and abundance related to substrate characteristics were more evident in the lower reaches of the estuary where sediments were more heterogeneous and localized patches of shell provided critical habitat for a variety of epifaunal organisms. The greater microhabitat complexity, combined with the higher and more stable salinities, undoubtedly contributed to the greater species richness in the harbor basin and lower reaches of the three rivers. This trend toward higher species richness progressing from the upper to the lower end of a relatively homoiohaline gradient estuary has been well documented by other researchers (Boesch, 1977a; Boesch *et al.*, 1976; Calder *et al.*, 1977; Dorges, 1977; Holland *et al.*, 1986, 1989). The importance of sediment type and spatial heterogeneity in determining benthic community structure has also been well established (Boesch, 1973; Bloom *et al.*, 1972; Carriker, 1967; Gray, 1974; Sanders, 1968; Thorson, 1966).

Temporal patterns of distribution and abundance were not as readily apparent as spatial trends. Although total numbers of individuals and species varied greatly throughout the four-year study, there was no consistent seasonal or annual periodicity in these fluctuations. This seemingly erratic temporal variability was characteristic of many numerically dominant species as well. These findings are consistent with those of other estuarine researchers (Boesch *et al.*, 1976; Flint and Kalke, 1985; Holland *et al.*, 1987, 1989). Boesch *et al.* (1976) observed that estuaries are typically inconstant and often unpredictable environments dominated by "r-strategists" which are characterized by their widely fluctuating abundances. These eurytopic opportunists, many of which were dominant in the Charleston Harbor estuary, are well adapted to exploiting polluted or disturbed habitats, by virtue of their high reproductive rates and short generation periods. This may explain the sporadically higher abundances of certain species (*e.g.*, *P. pinnata*, *S. benedicti*, and *M. lateralis*) subsequent to rediversion. Holland *et al.* (1989) observed a similar increase in the frequency and abundance of euryhaline marine species in response to subtle increases in salinity resulting from a regional drought.

Effects of rediversion on the benthos are difficult to infer from our study in the absence of a long-term pre-rediversion database with which to compare our results. The few ecological studies that have been conducted in the Charleston Harbor estuary provide very cursory surveys of the benthos that are not amenable to quantitative comparison with our study due to differences in sampling locations or methods (Academy of Natural Sciences, 1974; Calder *et al.*, 1977; Dames and Moore, 1975; Williams, 1984). Nevertheless, a review of these studies suggests that the qualitative composition of the macrobenthos has not changed markedly since rediversion. This finding is consistent with Boesch's (1974) observation that estuarine communities are resistant to change in face of environmental perturbations. Holland *et al.* (1989) drew a similar conclusion from their research in the Chesapeake Bay.

Despite the lack of evidence for drastic alterations of the benthos as a result of rediversion, a few relatively stenotopic species (*i.e.*, the amphipod *Lepidactylus dytiscus*, the bivalve *Petricola pholadiformis* and the polychaete *Scolecolepides viridis*) appeared to exhibit a trend toward decreasing abundance in certain reaches of the estuary subsequent to this event. Reduced freshwater inflow as a result of river diversion has been demonstrated to increase salinities, decrease riverborne nutrient inputs and aggravate contaminant problems in estuaries around the world (Benson, 1981; Funicelli, 1984; Hedgpeth and Rozengurt, 1985). A review of the long-term impact of stream diversion on the hydrological and biological features of several estuarine ecosystems indicated that a steady reduction of annual and seasonal runoff exceeding 30 and 40%, respectively, will result in the destruction of the dynamic equilibrium of estuaries within 3-7 years (Hedgpeth and Rozengurt, 1985). In light of the fact that the freshwater inflow to the Charleston Harbor estuary has been reduced by approximately 70% since the Cooper River was rediverted in 1985 (see Chapter III), and that our post-rediversion sampling extended only three years beyond this event, it may be that the benthic communities will exhibit greater changes related to rediverison in the future.

SHORT-TERM INTENSIVE STUDY

METHODS

Field and Laboratory:

Sampling for the intensive benthic assessment was conducted at 178 stations located throughout the harbor basin and the lower portions of each river system (Figure VII.23). Stations in the Ashley River extended to an area just below the North Bridge and encompassed the industrialized portion of that river. Stations in the Cooper River extended to Snow Point and encompassed most of the Naval port facilities located along that river as well as the industrial sites located in the lower portion of the river. Stations in Wando River extended to an area near the Highway 41 bridge which encompassed the port facilities in that river and a portion of the river above the port facilities that has not experienced much development.

The stations were separated by approximately 1-km intervals along the estuarine gradient in transects extending across the harbor basin and river channels from the 3-m depth contours to mid-channel depths to accommodate both depth and salinity gradients within the study area (Figure VII.23). During sampling, stations were located using fixed landmarks or Loran-C positioning where the use of fixed landmarks was not feasible. Loran-C coordinates were recorded for all sites. All sampling for this study was completed during a two-week period in July, 1988.

A single 0.05-m^2 Ponar grab sample was collected at each of the 178 stations for assessment of the benthic assemblages and sediment characteristics. Measurements of bottom salinity, temperature and dissolved oxygen were also taken at every site using a Hydrolab II Water Quality System. A small subsample of sediment was obtained from each grab for analysis of sediment composition by vertically inserting a 3.5-cm diameter core tube to the maximum depth of the grab sample (see Chapter VI for additional details). A second subsample (25 ml) of sediment was obtained from the surface of each grab sample for analysis of selected trace metals (see Chapter XI for further details). The remainder of the grab sample was then measured for sediment volume and sieved through a 0.5-mm screen. Benthic fauna retained on the sieve were preserved in a 10% seawater-formalin solution stained with rose bengal. The fauna were identified in the laboratory to species level or the lowest taxonomic level feasible and counted. Two taxonomic groups, Oligochaeta and Nematoda, were not identified to species level due to taxonomic difficulties with these groups, but these taxa were considered in the community analyses since they were abundant in many of the samples collected.

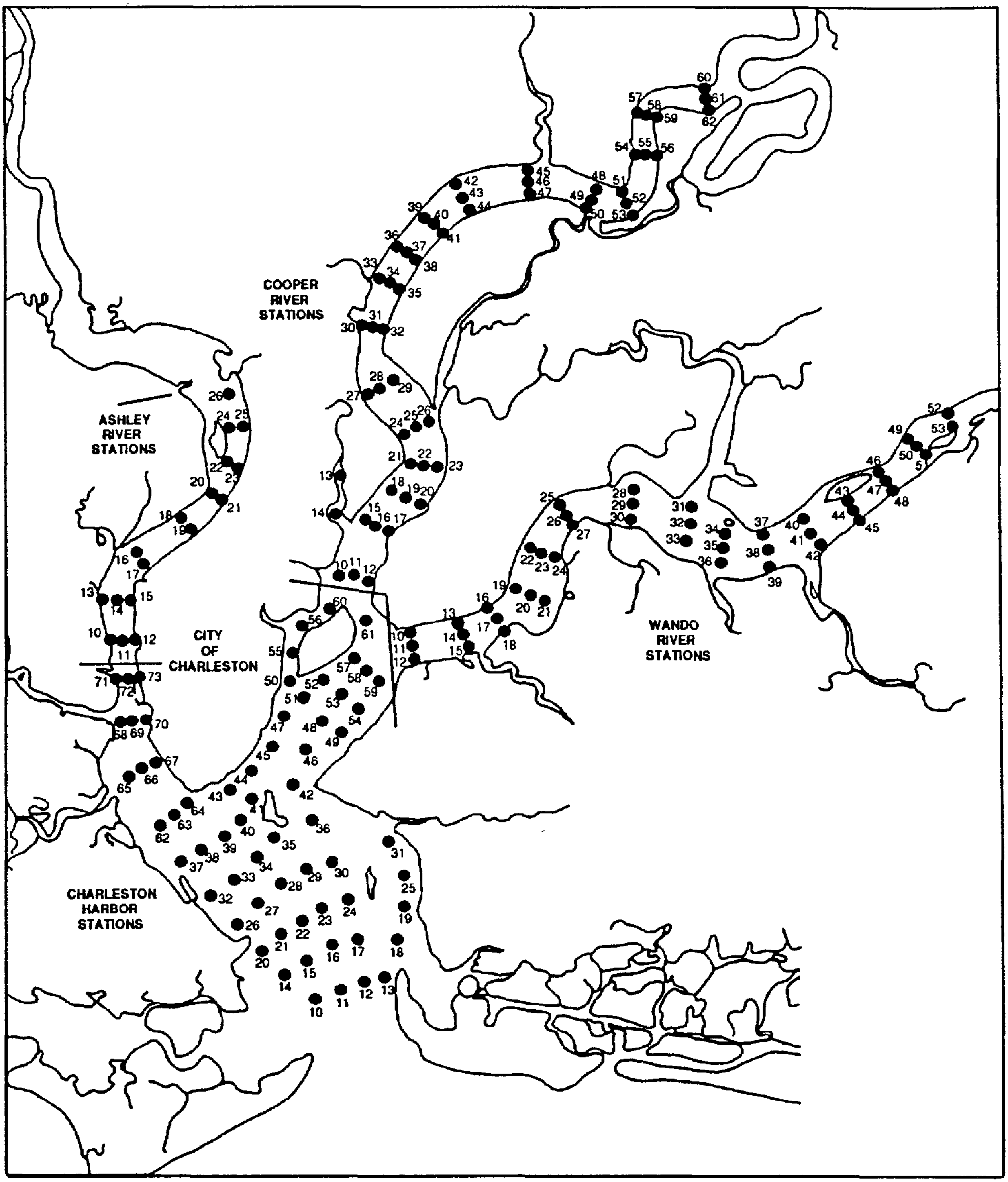


Figure VII.23. Location of stations sampled in July 1988 for the intensive assessment of sediments and benthic communities.

Data Analysis:

Benthic grab data were analyzed to determine the total faunal abundance and number of unique taxa present at each site, as well as to obtain estimates of species diversity (H'), evenness (J'), and species richness (SR) as described by Pielou (1975) and Margalef (1958). Species collected from all 178 stations were then ranked according to overall density and percent occurrence among the stations. Due to the large size of the station-by-species matrix, the data file was reduced to eliminate taxa which occurred at three or fewer stations, and to eliminate taxa which could only be identified to taxonomic categories due to specimen damage. The remaining data were then log-transformed [$\log_{10}(x + 1)$] to reduce the relative importance of exceptionally abundant taxa.

Principal component analyses were performed on the reduced data matrix, with and without orthogonal (varimax) rotations of the components. These analyses were conducted to determine the extent of colinearity among the environmental variables and to evaluate relationships between selected biological variables (abundance of numerically dominant species; total faunal abundance; number of species; and the diversity indices H', J', and SR), and selected environmental variables (station distance up the estuary; bottom depth; bottom water measurements of salinity, temperature, and concentrations of copper and chromium in the sediment).

The similarity of community composition among stations and the relationships between benthic community structure and the environmental variables measured were evaluated for the reduced data set (105 species by 178 stations) using the multi-variate Ecological Analysis Package (Ecoanalysis Inc., 1985) in conjunction with the Statistical Analysis System (SAS Institute, 1985). The following is a brief summary of the ordination and cluster analyses used; Smith et al. (1987) provides a more detailed description of these procedures and the rationale for their use. The Bray-Curtis index (Bray and Curtis, 1957; Clifford and Stephenson, 1975) was first used to obtain estimates of sample dissimilarities which were re-estimated using a step-across procedure (Williamson, 1978; Smith, 1984). Detrended correspondence analysis (Hill and Gauch, 1980; Gauch, 1982) and nonmetric multidimensial scaling (Kruskal, 1964a; 1964b; Kruskal and Wish, 1978) were used in conjunction with principal coordinates analysis (Gower 1966, 1967) to create initial and final configurations in ordination space, respectively. Euclidean distances calculated between sites in the final ordination space were used in an agglomerative, hierarchical clustering method called flexible sorting (Lance and Williams, 1967; Clifford and Stephenson, 1975) using the standard beta coefficient of -0.25. Both normal (station) and inverse (species) dendrograms were produced, along with a two-way coincidence table to relate species groups and station groups.

A weighted discriminant analysis was also performed on the reduced data set using the Ecological Analysis package (Ecoanalysis, Inc., 1985) in order to determine how the station groups resulting from the cluster analysis related to the environmental variables. This analysis produced coefficients of separate determination which were used to indicate which environmental variables were important in determining the position of a sample on the respective axis. Plots of the station groups in multidimensional space were also produced from this analysis.

Prior to performing the weighted discriminant analysis, a subset of environmental variables was selected from all those measured based on the initial principal components analysis and preliminary multiple regression analyses using the RSQUARE procedure (SAS, 1985). These analyses showed (1) no correlation between the environmental variables, temperature and dissolved oxygen, and the biological variables considered; (2) high colinearity among the sediment variables percent sand, percent silt and percent clay contents, and between percent organic content and the concentrations of copper and chromium in the sediment; and (3) a high correlation between station distance from the harbor entrance and bottom salinity. Therefore, subsequent multiple regression analyses of the community ordination scores using the RSQUARE procedure were limited to the following environmental variables: distance from the harbor entrance (kilometer), bottom depth, percent sand, mean grain size of the sand fraction, percent organic content, and percent $CaCO_3$ content. Station distance was used to indicate the station's relationship to a general salinity regime. This variable was selected instead of bottom salinity since field collections were not standardized to a particular tidal stage and salinities varied considerably over a tidal cycle in this estuary.

All percentages were arcsine transformed to correct for non-normality of the data and heterogeneous variances. A coefficient of determination (R^2) was computed for each combination of the environmental variables using the RSQUARE procedure to determine which environmental explained most of the variance in the formulation of regression models.

RESULTS AND DISCUSSION

Community Diversity and Abundance Patterns:

Grab samples obtained from the 178 stations contained a total of 22,596 organisms representing 185 taxa (Table VII.13). A listing of the species present at each station is provided in Appendix VII.B. This Appendix also provides data on the total abundance, number of species and estimates of the diversity parameters H', J', and SR for each station.

Table VII.13. Total number of organisms and taxa collected at 178 stations sampled in Charleston Harbor during July, 1988.

Taxon	No. of Organisms	No. of Species
Pelecypoda	10,060	25
Polychaeta	8,297	65
Oligochaeta	1,521	*
Amphipoda	811	28
Nematoda	465	*
Gastropoda	345	19
Nemertinea	265	*
Isopoda	191	7
Decapoda	175	18
Mysidacea	98	1
Actiniaria	91	1
Ostracoda	78	2
Cumacea	69	4
Other Taxa	130	15*
Total	22,596	185

* Taxon includes organisms not identified to the species level, actual number of species unknown.

Figures VII.24-VII.26 depict a spatial representation for three of these parameters: species diversity (H'), total number of species, and total faunal abundance throughout the study area.

The faunal abundance and diversity estimates showed considerable variability among the stations sampled (Appendix VII.B) and a principal components analysis indicated that these community parameters were not strongly correlated with any of the environmental variables measured ($r \leq 0.43$, Table VII.14). Weak negative correlations were noted between station distance from the harbor entrance and estimates of H', SR and the $\log_{10}$ number of species. These biological variables also showed weak positive relationships with bottom salinities and the $CaCO_3$ content of the sediments. However, a subsequent regression

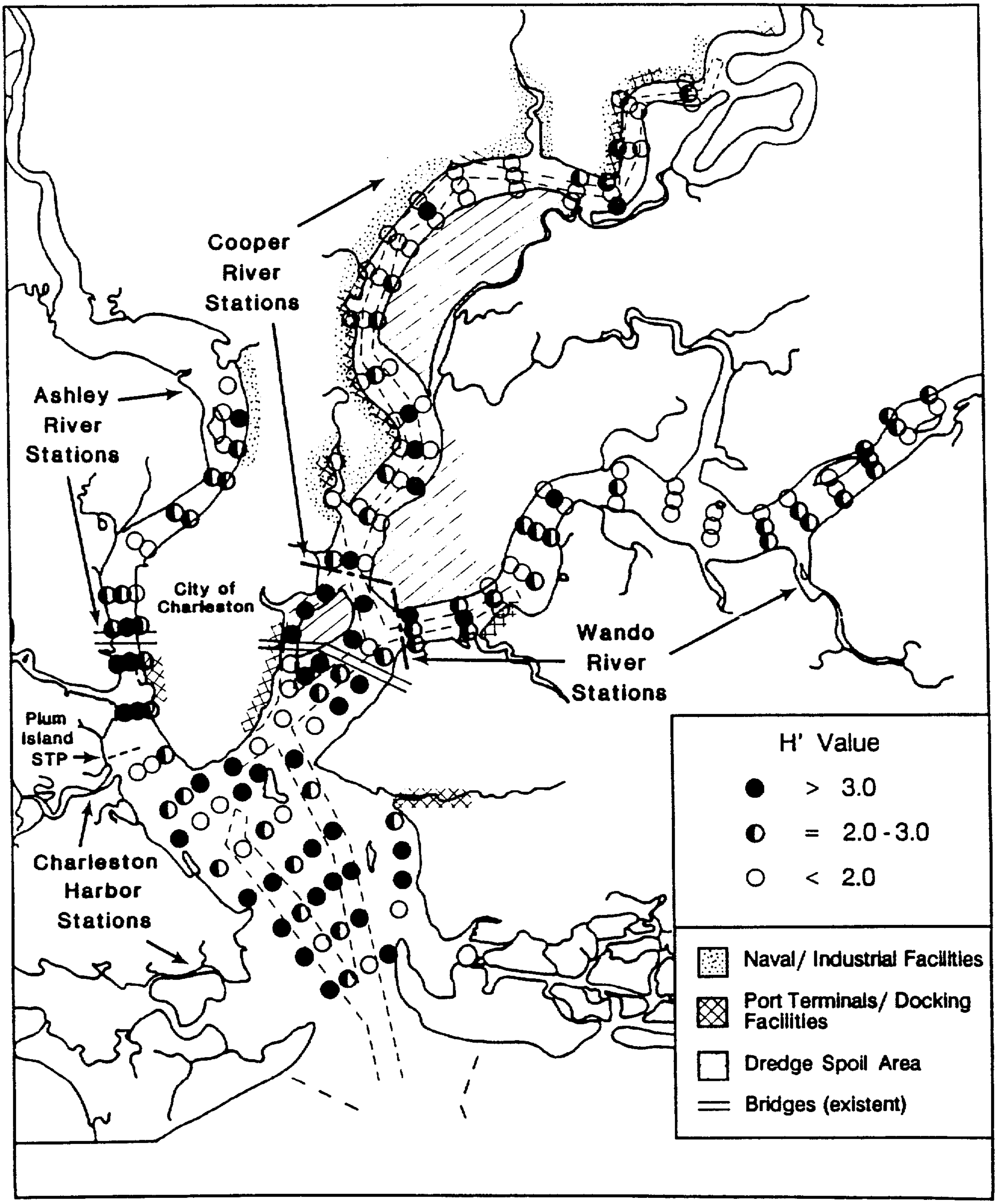


Figure VII.24. Diversity values (H') for the benthic communities sampled at the 178 stations in July, 1988.

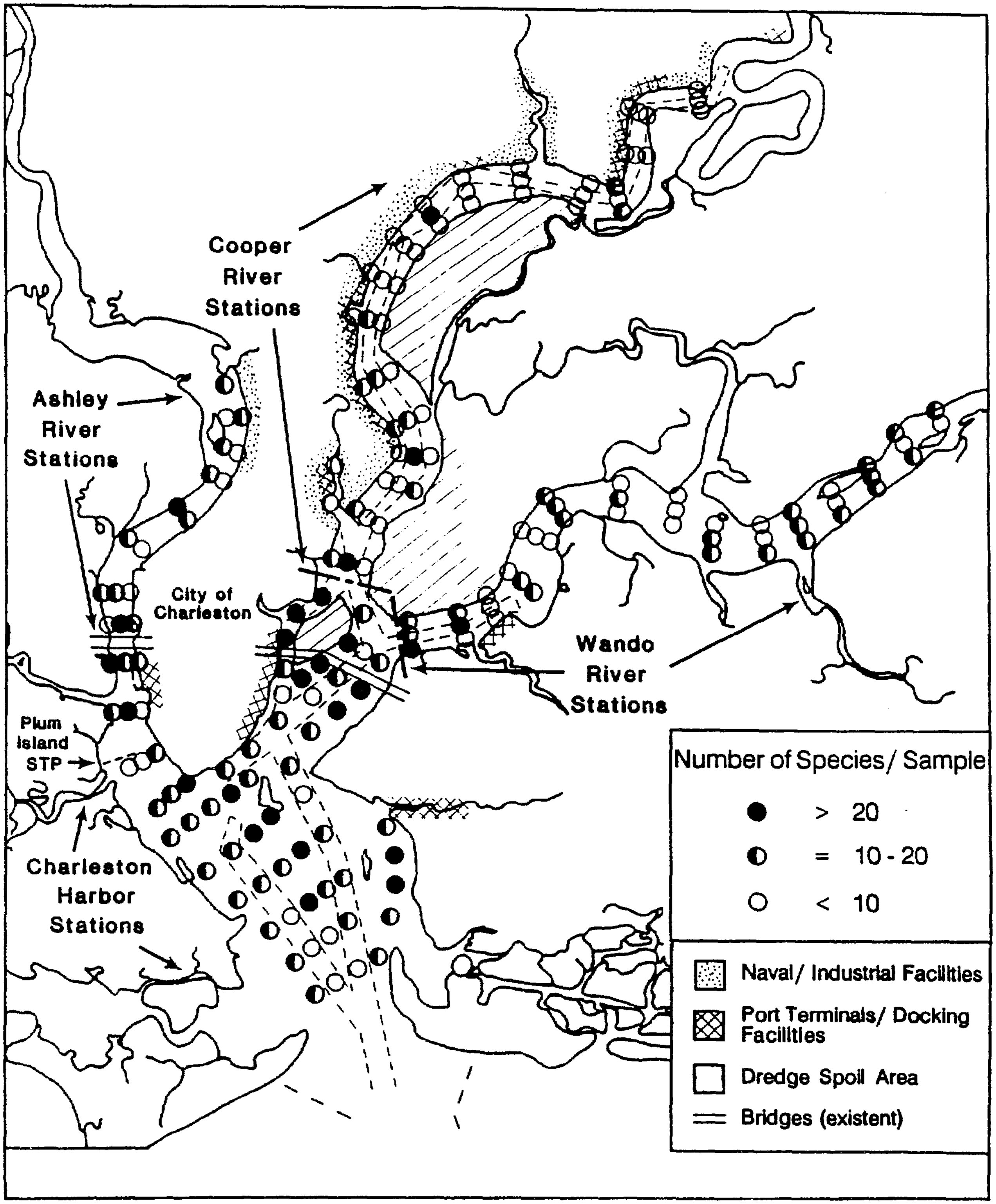


Figure VII.25. Number of benthic macrofaunal species collected at the 178 stations sampled in July, 1988.

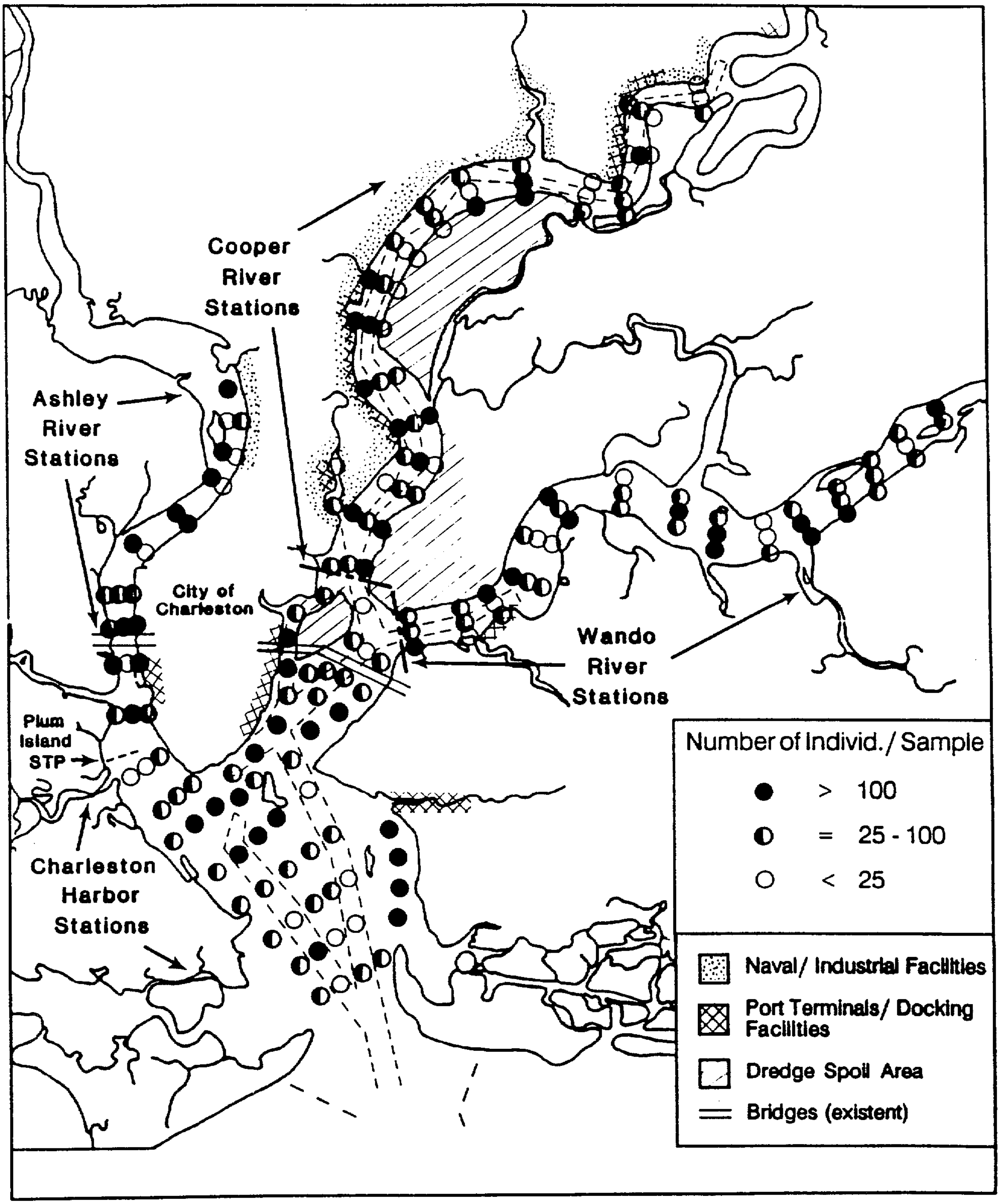


Figure VII.26. Abundance of benthic macrofauna collected at the 178 stations sampled in July, 1988.

Table VII.14. Pearson produce-moment correlations obtained from principal components analysis of selected environmental and biological variables.

BIOLOGICAL VARIABLES	ENVIRONMENTAL VARIABLES												
	Station Distance	Station Depth	Bottom Salinity	Bottom DO	Bottom Temp.	% Sand	% $CaCO_3$	% Silt	% Clay	% Organic	x Phi	Copper	Chromium
Dominant Species:													
Mulinia lateralis	-0.253	-0.018	0.185	0.257	0.016	-0.311	-0.011	0.405	0.067	0.327	0.380	0.399	0.299
Paraprionospio pinnata	0.153	-0.078	-0.055	-0.163	0.092	-0.321	-0.168	0.307	0.261	0.406	0.219	0.390	0.374
Heteromastus filiformis	-0.003	-0.177	-0.018	0.093	0.248	-0.173	0.030	0.193	0.053	0.162	0.221	0.105	0.113
Nereis succinea	0.243	-0.095	-0.258	-0.084	0.097	-0.150	0.059	-0.088	0.242	0.056	-0.006	-0.037	0.053
Spiochaetopterus costarum oculatus	0.010	-0.145	0.032	0.107	0.221	0.021	0.155	-0.051	-0.070	-0.098	0.074	-0.092	-0.120
Glycinde solitaria	-0.496	0.117	0.448	0.233	0.080	-0.088	0.275	0.118	-0.148	-0.023	0.188	-0.071	0.037
Clymenella torquata	-0.342	0.098	0.286	0.222	0.034	0.072	0.365	-0.151	-0.175	-0.168	0.161	-0.197	-0.108
Melita nitida	0.122	-0.003	-0.050	-0.048	0.015	-0.013	0.288	-0.093	-0.028	-0.057	-0.147	-0.125	-0.059
Streblospio benedicti	-0.031	0.029	-0.001	0.046	0.071	-0.133	0.162	0.012	0.099	0.083	0.142	0.127	0.092
Ilyanassa obsoleta	-0.005	-0.186	-0.069	0.125	0.106	-0.109	-0.063	0.165	0.047	0.140	0.036	0.192	0.141
Community Variables:													
Total Number of Organisms/Sample	-0.181	-0.030	0.207	0.171	0.136	-0.353	0.121	0.303	0.140	0.229	0.357	0.036	0.276
Total Number of Species/Sample	-0.432	0.015	0.391	0.336	0.144	-0.143	0.400	0.078	-0.104	-0.029	0.256	-0.064	0.036
Species Diversity (H')	-0.377	0.070	0.317	0.230	0.016	0.080	0.358	-0.130	-0.196	-0.222	0.056	-0.232	-0.128
Evenness (J')	-0.129	0.086	0.094	0.026	-0.068	0.190	0.156	-0.194	-0.171	-0.239	-0.129	-0.230	-0.173
Species Richness (SR)	-0.413	0.040	0.344	0.300	0.062	0.011	0.434	-0.087	-0.178	-0.181	0.126	-0.220	-0.100

analysis of the $\log_{10}$ number of species against the two best correlates for this variable (station distance and percent $CaCO_3$) indicated that these environmental parameters explained only 36% of the regression model (R^2 =0.36). Similarly, regression analysis of the $\log_{10}$ number of individuals/station with the best correlates for that variable (percent sand and sand phi) resulted in an even lower R^2 coefficient (0.16).

The weak relationships noted between these general community parameters and station location in the estuary may have been stronger if sampling had been extended further up the estuary to include more brackish waters. Our study sites were primarily restricted to the polyhaline zone of this estuary. Carriker (1967), Boesch (1977b), Holland *et al.* (1989) and others have noted that estuarine systems typically have lower macrobenthic diversities in brackish water habitats compared with more saline environments. The apparent relationship between benthic diversity and $CaCO_3$ content of the sediments would also be expected since increased shell material in the sediments should provide a more spatially complex substratum.

Within the harbor basin, relatively low diversities were noted at a few channel stations near the harbor entrance, at several sites near the mouth of the Ashley River, and at many of the sites located along the city shoreline adjacent to port facilities (Figure VII.24). Samples from three of these sites, (CH58, 65, 66) contained very few organisms representing only a few species (Figures VII.25, VII.26), even though the sample volumes indicated adequate grab penetration into the sediments. Low diversities observed at the other sites were attributable to low evenness values resulting from high abundances of one or a few species, especially the bivalve *Mulinia lateralis*. The average H' value among the harbor basin (CH) stations was 2.77 bits/individual.

No clear patterns of diversity were observed along the portions of the three river systems sampled (Figure VII.24). However, significant differences were noted in the H' values observed among the rivers ($P<0.05$, Kruskal-Wallis test). The Cooper River had the lowest average H' values (mean H'= 1.769) as well as the greatest percentage of stations with H' values less than 2.0 bits/individual (Figure VII.24). Average H' values in the Ashley and Wando rivers were 2.37 and 2.05 bits/individual, respectively. The low diversity values in all three river systems were usually attributable to low species richness (Appendix VII.B). The proximity of many stations to sources of anthropogenic stress in the harbor basin and river systems may account for some of the diversity patterns observed (see later discussion), but these relationships are not well-defined.

Dominant Taxa:

Pelecypods were the most numerous benthic organisms collected in the estuary, due largely to the abundance of one species, *Mulinia lateralis*. This species accounted for 96.5% of all bivalves collected and 43% of the total faunal abundance from all sites combined (Table VII.15). Although *M. lateralis* were present throughout the study area, their distribution was highly contagious ($P<.01$, x^2 test). Greatest densities of *M. lateralis* were observed in the lower harbor basin and in the Cooper River (Figure VII.27). Principal components analysis indicated weak positive correlations between the abundance of this

Table VII.15. Relative abundance and frequency of occurrence of taxa present at 10% or more of the 178 benthic stations sampled in July, 1988. A = Amphipoda, C = Cumacea, D = Decapoda, I = Isopoda, M = Mollusca, N = Nemertina, P = Polychaeta.

Taxa	Total Number	% of Total Abundance	# of Stations W/Organisms	% of Total Stations
Mulinia lateralis (M)	9712	43.0	86	48.3
Paraprionospio pinnata (P)	4554	20.2	129	72.5
Oligochaeta	1521	6.7	124	69.7
Heteromastus filiformis	1018	4.5	103	57.9
Nematoda	465	2.1	48	27.0
Nereis succinea (P)	401	1.8	61	34.3
Spiochaetopterus costarum oculatus (P)	371	1.6	48	27.0
Glycinde solitaria (P)	345	1.5	87	48.9
Clymenella torquata (P)	247	1.1	45	25.3
Melita nitida (A)	199	0.8	22	12.4
Nemertina	164	0.7	82	46.1
Streblospio benedicti (P)	147	0.7	39	21.9
Carinomella lactea (N)	101	0.5	40	22.5
Batea catharinensis (A)	94	0.5	23	12.9
Scoloplos rubra (P)	91	0.4	28	15.7
Tellina texana (M)	90	0.4	29	16.3
Paracaprella tenuis (A)	87	0.4	22	12.4
Capitella capitata (P)	85	0.4	18	10.1
Ostracoda	78	0.4	33	18.5
Edotea montosa (I)	71	0.3	39	21.9
Aligena elevata (M)	68	0.3	18	10.1
Leucon americanus (C)	65	0.3	23	12.9
Cirratulidae undet. (P)	63	0.3	26	14.6
Glycera americana (P)	61	0.3	38	21.4
Diopatra cuprea (P)	49	0.2	20	11.2
Panopeus herbstii (D)	41	0.2	20	11.2
Astyris lunata (M)	40	0.2	19	10.7
Sigambra tentaculata (P)	38	0.2	21	11.9
Phyllodoce arenae (P)	20	0.1	18	10.1

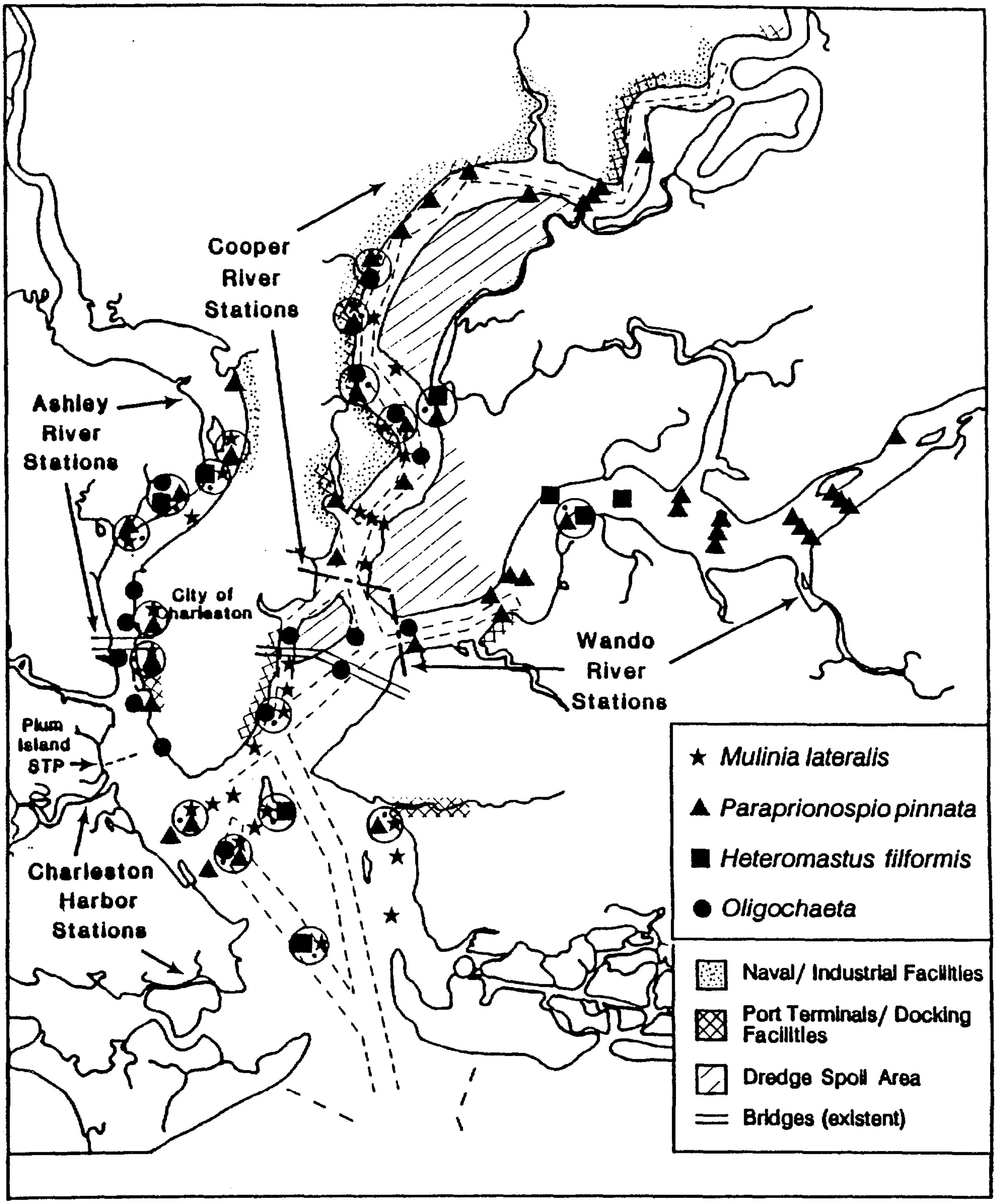


Figure VII.27. Benthic samples which had greater than 500 individuals/m² of at least one of the three most abundant species. Stations with circles had high abundances of two or more of these species, with the station location represented by a small dot.

species and percentages of silt and organic matter in the sediments. Twenty-three other species of bivalves were collected at the 178 stations sampled, but only *Tellina texana* was present at more than 10% of the stations and no other bivalve species contributed to more than 1% of the total faunal abundance (Table VII.15).

Polychaetes formed the second most abundant taxonomic group. These organisms were also the most diverse with respect to the number of known species represented in the samples (Table VII.13). Among the 65 species identified, 14 were found at more than 10% of the stations and 6 species each accounted for more than 1% of the total faunal abundance from all sites combined (Table VII.15). These species were *Paraprionospio pinnata, Heteromastus filiformis, Nereis succinea, Spiochaetopterus costarum ocalatus, Glycinde solitaria*, and *Clymenella torquata*. All of these polychaetes were present throughout most of the study area; however, *P. pinnata, N. succinea*, and *S. costarum ocalatus* were somewhat more abundant in the middle reaches of the estuary (upriver extent of our sampling area), whereas *G. solitaria* and *C. torquata* were more abundant in the harbor basin and lower reaches of the rivers. None of these species exhibited distribution patterns that were highly correlated with the environmental variables measured (Table VII.14).

Other taxonomic groups which were relatively abundant were Oligochaeta, Nematoda, Amphipoda, Gastropoda, and Decapoda (Table VII.13). Oligochaetes were found at approximately 70% of the stations and represented 6.7% of all taxa collected by number. Nematodes were present at 27% of the stations, but contributed to only 2.1% of the total abundance. None of the species within the latter three taxonomic groups contributed more than 1% of the total faunal abundance, although several species were present at more than 10% of the stations sampled (Table VII.15). These included the amphipods *Melita nitida, Batea catharenensis*, and *Paracaprella tenuis*; the bivalve *Aligena elevata*; the gastropod *Astyris lunata*; and the mud crab *Panopeus herbstii*.

General Community Composition and Distribution Patterns:

Ordination analysis of the reduced 105 species by 178 stations matrix indicated extreme variability in the data set, with 10 axes required to explain 93.5% of the total variance in ordination space. The first three axes accounted for approximately 48% of the variance explained, with the remaining axes each accounting for 10% or less of the total variance. Plots of the station groups derived from the cluster analysis are presented for these first three axes in Figure VII.28 and were used to further interpret the relative similarity among station groups based on the biological patterns observed. Dendrograms generated by the normal and inverse cluster analyses are presented in Figures VII.29 and VII.30 along with a two-way coincidence table which summarizes the relative percent

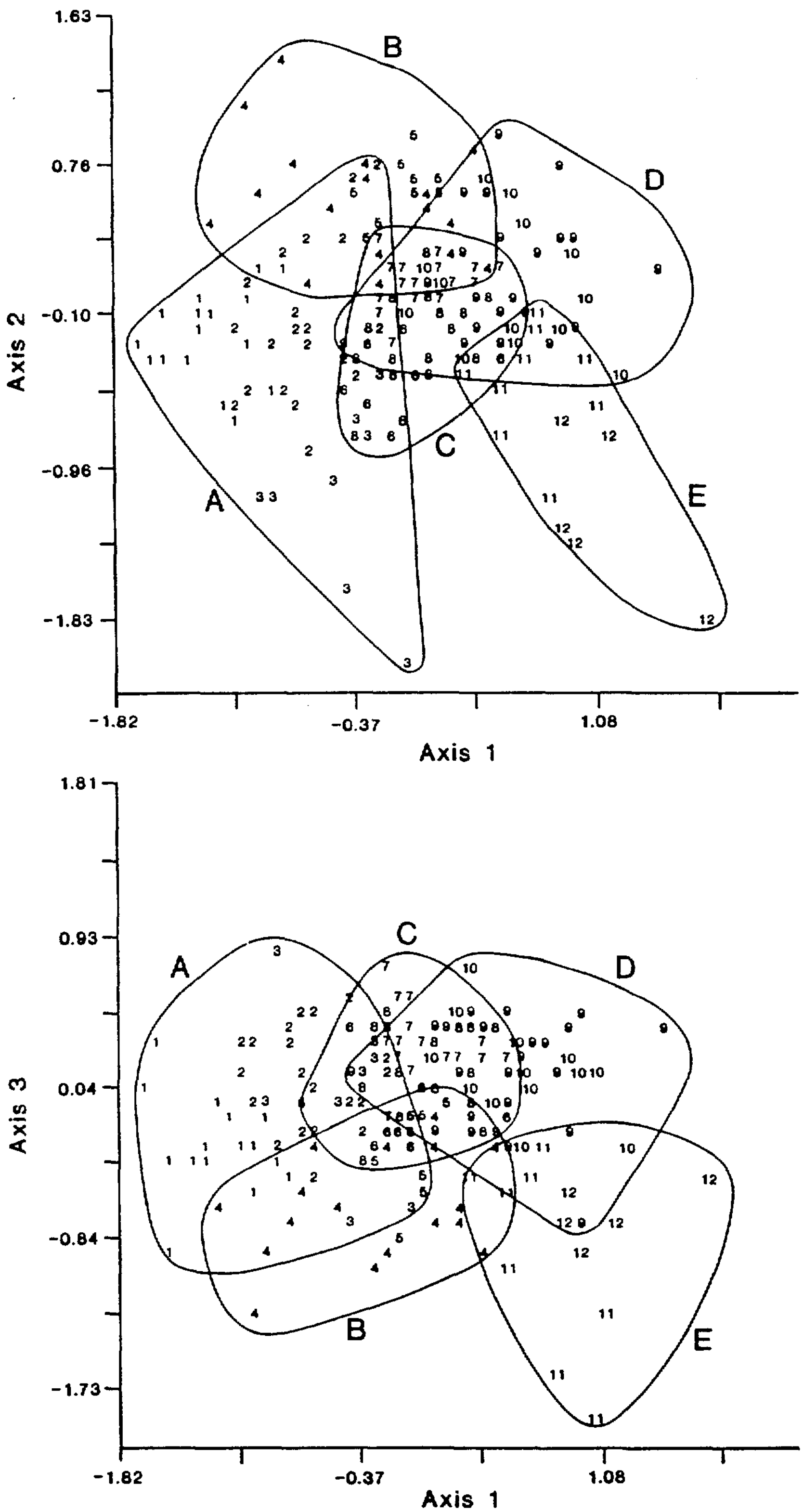


Figure VII.28. Distribution of stations along the first three axis of principal components ordination space. Station numbers denote station groups defined by normal cluster analysis. The larger station groups A-E are outlined in each graph. There are 21 overlapping groups in the upper figure and 23 overlapping groups in the lower figure.

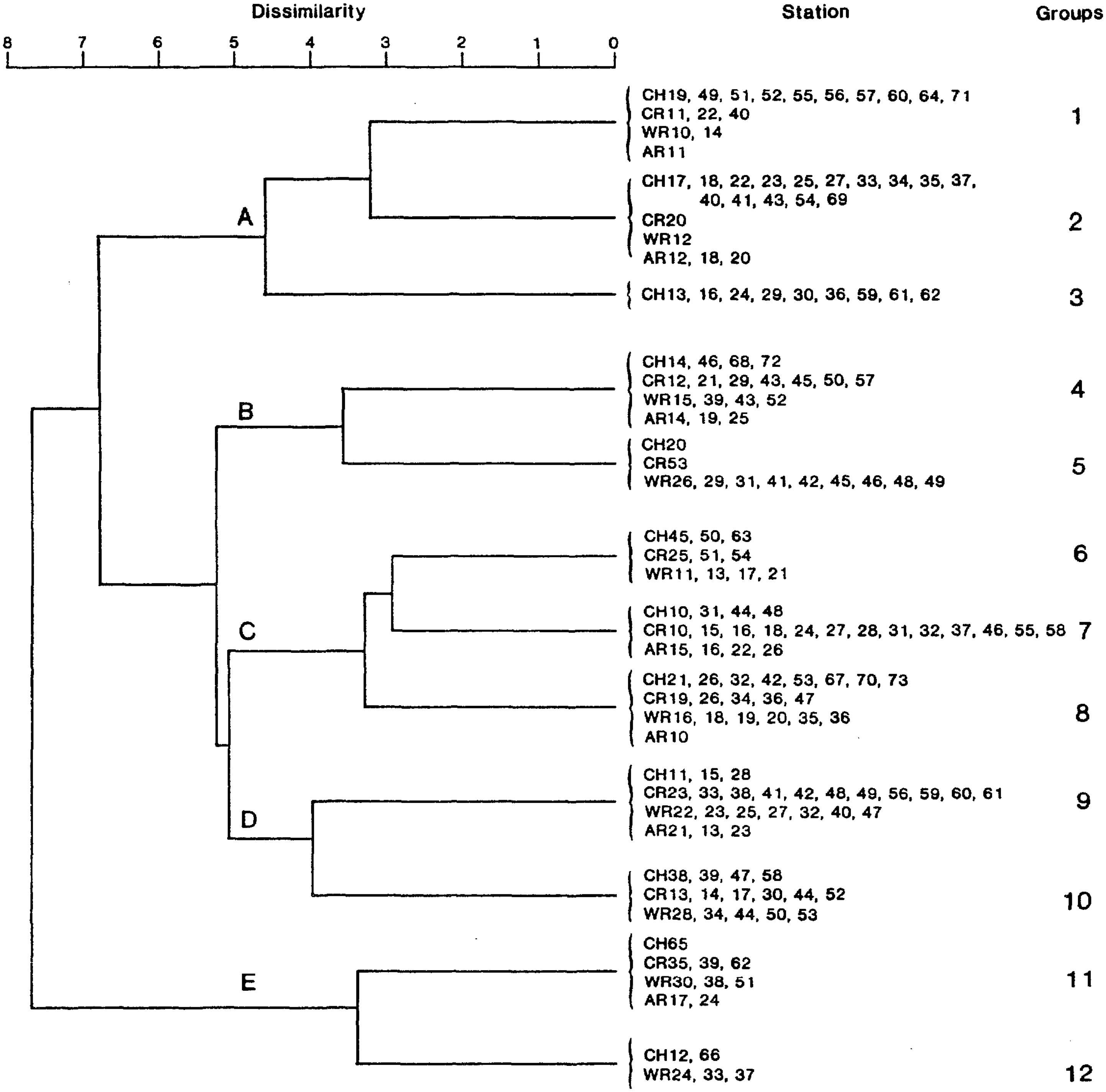


Figure VII.29. Summary dendrogram of station groups resulting from the normal cluster analysis of benthic macrofauna sampled at the 178 grab stations.

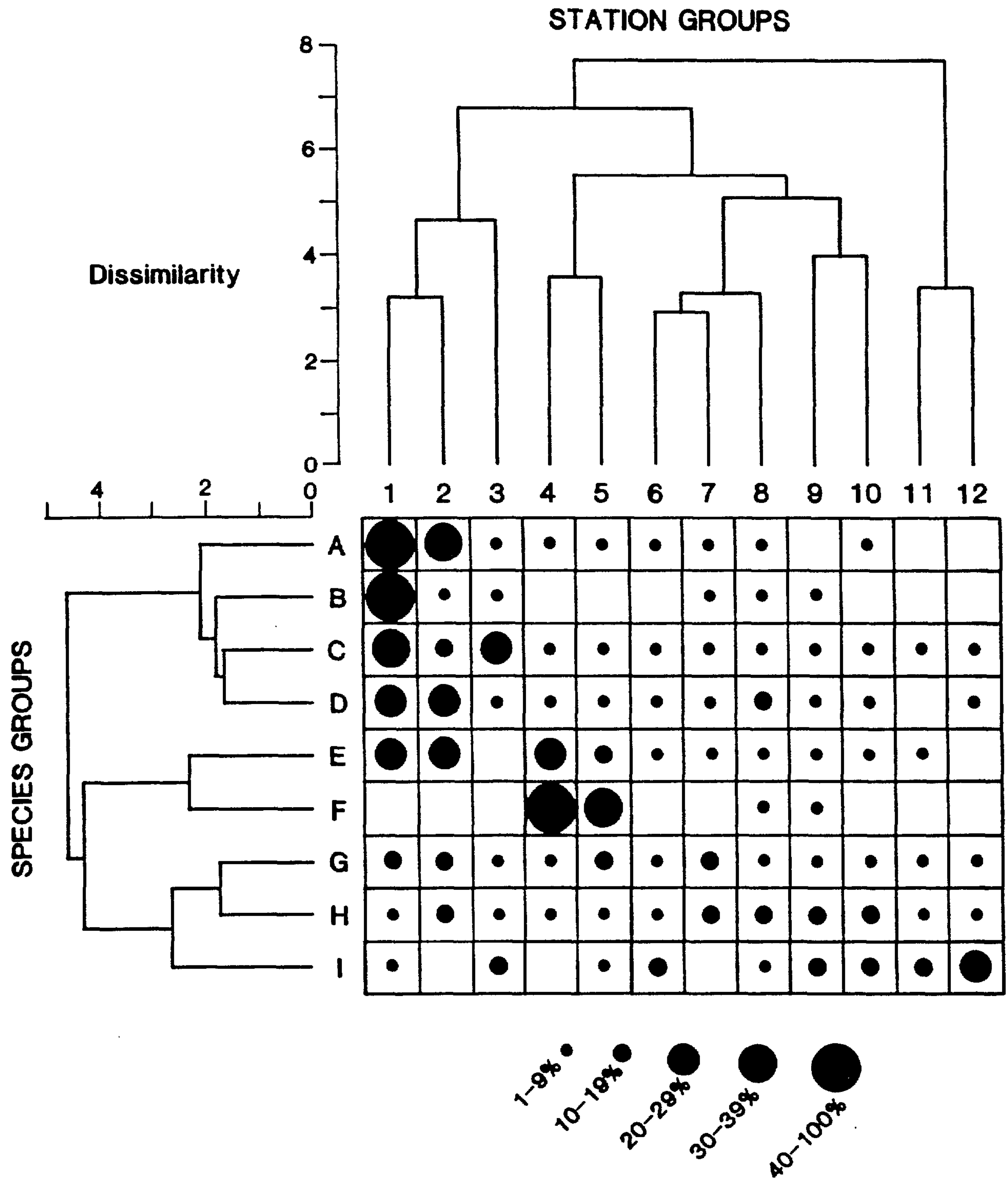


Figure VII.30. Two-way classification table showing the normal and inverse cluster dendrograms and the percent occurrence of each species group in each station group. Row percentages sum to 100%.

occurrences of each species group in each station group. A list of the species in each of the inverse cluster groups, and a geographical plot of the station groups generated by the normal cluster analysis appear in Table VII.16 and Figure VII.31, respectively.

The normal cluster analysis identified 12 station groups which were contained within five larger groups (A-E, Figure VII.29). The first large group (A) was comprised of three station subgroups (1-3), which were mostly located in the lower portion of the estuary (Figures VII.29, VII.31). In fact, only about 10% of the river sites (CR, WR, AR) were in this larger group. Species commonly found at many of these stations were those in species groups A-D (Figure VII.30, Table VII.16). Many of these species included epifaunal and infaunal organisms which are typically present in poly-and euhaline habitats with sand and muddy sand bottoms (Bousfield, 1973; Day, 1973; Abbott, 1974; Gardiner, 1975; Van Dolah *et al.*, 1979, 1983, 1984a, 1984b; Williams, 1984; Ruppert and Fox, 1988; Winn *et al.*, 1989), similar to those characteristic of stations in Group A (Table VII.17). Ordination analyses showed relatively little separation of station subgroups 1-3, although the one station subgroup (3) was moderately separated from the others along axis 2 (Figure VII.28). Stations in subgroup 3 generally had a higher sand content and were found lower in the estuary than the stations in subgroups 1 and 2.

The second large station group (B) contained two subgroups of stations (4 and 5) which were scattered throughout the study area (Figures VII.29, VII.31). Taxa most commonly found at these stations were those in species group F, and to a lesser extent, species group E (Figure VII.30). Many of the species in these groups were epifaunal organisms commonly associated with sandy bottoms and firmer substrata (Bousfield, 1973; Abbott, 1974; Gardner, 1975; Ruppert and Fox, 1988). Sediments at the stations in subgroup 4 were predominantly sandy with a relatively high clay content (Table VII.17). Stations in subgroup 5, on the other hand, were predominantly sandy with relatively little silt and clay. Both of these station groups showed considerable overlap in ordination space and they were not well separated from any of the other major station groups (Figure VII.28).

Stations in groups 6-10 comprised the majority of sites sampled in the three river systems. The stations in these five groups formed two larger groups (C and D), which showed greater similarity to each other than to all other station groups (Figure VII.29). These groups also showed considerable overlap with each other in ordination space. Sediments at most of the stations in these groups were generally a mix between sands and fine materials (Table VII.17). None of the species groups identified in the inverse cluster analysis showed a strong affinity to station subgroups 6-10.

Table VII.16. Species groups resulting from inverse cluster analysis of the 178-station-by-105-species matrix analyzed using the Bray-Curtis similarity coefficient. (A = Amphipoda, An = Anthozoa, As = Ascidean, C = Cumacea, D = Decapoda, I = Isopoda, M = Mollusca, P = Polychaeta, S = Stomatopoda). Species with an * were present at more than 10% of the stations.

Group A

- *Neopanope sayi* (D)
- *Elasmopus levis* (A)
- *Unicola serrata* (A)
- *Pista palmata* (P)
- *Anadara transversa* (M)
- *Cyathura burbancki* (I)
- *Abra aequalis* (M)
- *Cerapus tubularis* (A)
- * *Batea catharinensis* (A)
- *Costoanachis avara* (M)
- *Lepidonotus sublevis* (P)
- *Dulichiella appendiculata* (A)
- *Notomastus lobatus* (P)
- *Corophium acherusicum* (A)
- * *Diopatra cuprea* (P)
- * *Aligena elevata* (M)
- * *Clymenella torquata* (P)
- *Portunus gibbesii* (D)
- *Lumbrineris tenuis* (P)
- * Ostracoda

Group B

- *Mercenaria mercenaria* (M)
- Turbonilla sp. (M)
- *Listriella clymenellae* (A)
- *Pseudeurythoe ambigua* (P)

Group C

- *Renilla reniformis* (An)
- *Spiophanes bombyx* (P)
- *Drilonereis magna* (P)
- *Listriella barnardi* (A)
- *Parvilucina multilineata* (M)
- *Turbonilla* sp. G (M)
- *Cistenides gouldii* (P)
- *Owenia fusiformis* (P)
- *Nucula proxima* (M)
- * *Tellina texana* (M)
- *Acteocina canaliculata* (M)
- *Drilonereis longa* (P)
- *Pinnixa chaetopterana* (D)
- *Leptochela serratorbita* (D)
- *Notomastus latericeus*

Group D

- *Magelona* sp. (P)
- * *Carinomella lactea* (N)
- * *Glycera americana* (P)
- * *Glycinde solitaria* (P)
- * *Sigambra tentaculata* (P)
- * Cirratulidae (P)
- *Mediomastus californiensis* (P)
- Sigalionidae (P)

Group E

- Turridae (M)
- *Petricola pholadiformis* (M)
- *Corophium simile* (A)
- * *Nereis succinea* (P)
- *Sabella microphthalma* (P)
- *Corophium tuberculatum* (A)
- *Lyonsia hyalina* (M)
- *Alpheus heterochaelis* (D)
- *Corophium lacustre* (A)
- *Erichthonius brasiliensis* (A)
- * *Panopeus herbstii* (D)
- *Sabellaria vulgaris* (P)
- * *Paracaprella tenuis* (A)
- Actiniaria
- *Palaemonetes vulgaris* (D)
- * *Melita nitida* (A)

Group F

- Muricidae (M)
- *Molgula manhattensis* (As)
- *Rhithropanopeus harrisii* (D)
- *Cyathura polita* (I)

Group G

- *Amygdalum papyrium* (M)
- * *Capitella capitata* (P)
- * *Spiochaetopterus costarum oculatus* (P)
- Phoronida
- * *Streblospio benedicti* (P)
- *Scolecolepides viridis* (P)
- Muricidae A (M)
- *Polydora socialis* (P)
- *Ampelisca vadorum* (A)
- * *Edotea montosa* (I)
- *Ampelisca abdita* (A)
- *Boonea impressa* (M)
- * *Phyllodoce arenae* (P)
- Turbellaria
- * *Scoloplos rubra* (P)
- *Trachypenaeus constrictus* (D)
- *Glycera dibranchiata* (P)
- *Pagurus longicarpus* (D)
- * Nematoda
- * *Astyris lunata*

Group H

- * *Leucon americanus* (C)
- * *Mulinia lateralis* (M)
- * Nemertinea
- * *Paraprionospio pinnata* (P)
- * *Heteromastus filiformis* (P)
- * Oligochaeta
- Squilla empusa (S)
- *Eteone heteropoda* (P)
- *Podarkeopsis levifuscina* (P)
- *Ilyanassa obsoleta* (M)
- Mysidacea
- *Leitoscoloplos* sp. (P)
- *Ogyrides alphaerostris* (D)
- *Monoculodes* sp. A (A)
- *Lepidactylus dytiscus*

Group I

- *Chiridotea almyra* (I)
- *Nephtys picta* (P)
- *Nephtys bucera* (P)

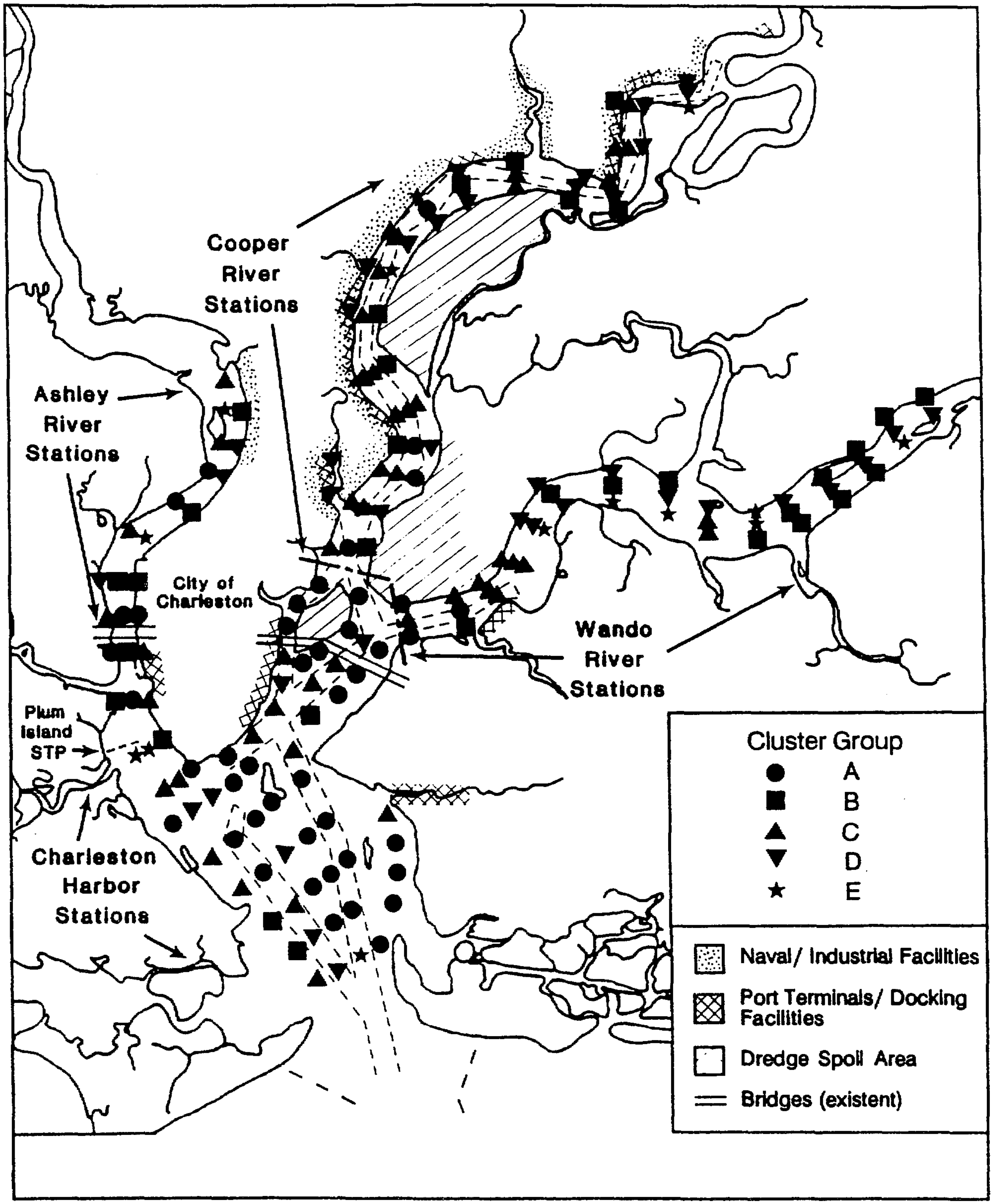


Figure VII.31. Geographical summary of the five major stations groups (A-E) defined by the normal cluster analysis.

Table VII.17. Summary of environmental characteristics for the twelve station groups defined by the normal cluster analysis.

Station Group	Station Subgroup	Average Depth (m)	Av. Distance from Harbor Entrance (km)	Av. Bottom Salinity (ppt)	Av. Bottom DO (mg/l)	Averages of Percent Sediment Type					Av. Sand Grain-Size (ϕ)
						Sand	Silt	Clay	$CaCO_3$	Organic	
A	1	9.2	9.3	27.7	5.1	61.2	6.8	16.6	15.4	3.7	2.4
	2	6.9	6.0	28.4	5.4	50.7	19.7	19.3	10.4	6.4	2.8
	3	10.1	4.2	30.5	5.4	79.8	5.1	7.9	7.1	2.2	2.4
B	4	7.4	15.1	23.7	5.1	46.3	11.5	39.2	3.0	7.1	2.3
	5	5.7	18.1	23.8	4.8	75.8	2.7	13.6	7.8	2.6	2.0
C	6	10.0	12.9	26.1	4.9	50.9	17.2	22.4	9.4	6.3	2.4
	7	8.7	14.0	25.1	5.0	40.1	24.3	31.1	4.5	8.7	2.8
	8	9.0	11.3	26.6	4.8	41.4	30.6	24.7	8.7	3.4	2.4
D	9	7.0	16.8	22.5	4.8	68.8	5.8	20.5	5.0	4.0	2.3
	10	7.5	15.1	25.3	4.7	55.5	18.9	21.1	4.4	6.7	2.4
E	11	5.1	17.5	21.7	4.8	83.5	2.3	11.8	2.5	3.0	1.3
	12	7.0	11.1	26.9	4.9	91.7	0.2	2.9	5.1	0.8	0.4

Station groups 11 and 12 were relatively dissimilar to all other station groups in the normal cluster analysis (Figure VII.29). These station groups were also moderately separated from the other stations groups in ordination space. None of the species groups showed a strong affinity for station group 11, and only species group I showed a moderate affinity to station group 12. Bottom sediments at stations in both of these groups were medium to coarse sands with a relatively low percentage of fine material (Table VII.17).

Results of the multiple regression analysis comparing the ordination scores with the environmental variables indicated that none of the variables measured were strongly correlated with the scores on any of the first three axes (Table VII.18). Weak relationships were noted between the biological patterns and the location of stations along the salinity gradient sampled as well as with several sediment variables, including percent sand, percent $CaCO_3$, percent organic content and sand-phi size. However, the three most important environmental variables measured explained only 35%, or less, of the variance on any of the axes and addition of the other variables did not improve the R^2 values significantly (Table VII.18).

The weighted discriminant analysis generally supported results obtained from the regression analysis (Table VII.19). The coefficients of separate determination for axis 1, which accounted for the greatest variance, indicated that station distance (which is indicative of the station's salinity regime) was the most important variable in separating the biologically defined site groups. Percent $CaCO_3$ also showed a higher relationship with the site group ordination scores than the other variables measured. Environmental variables most important on axis 2 were the mean grain size of the sand fraction, percent sand, and percent organic material. These same variables also had relatively high coefficients on axis 3. However, plots of the station groups on the three discriminant axes (not presented) indicated almost total overlap of most station groups on axes 2 and 3 and many of the station groups showed considerable overlap on axis 1, as well.

The lack of strong relationships between the ordination scores and the environmental variables measured in both the regression and discriminant analyses supports the patterns noted for the more general diversity and abundance estimates. As noted previously, the portion of the estuary sampled in this study segment was limited to the lower polyhaline areas where the majority of the shipping and industrial facilities are located. As a result, we did not observe as clear a distinction in benthic community composition related to the salinity gradient as was noted in the four-year study segment (see previous section) where a greater portion of the estuarine gradient was sampled. Similarly, although sediment characteristics varied considerably among the stations sampled, sands were the predominant component at most of the stations (69%), and only a few sites (12%) sampled had

Table VII.18. Results of the multiple regression variable selection technique using the ordination axis scores obtained from 178 stations by 105 species matrix as the dependent variable. Only the first three axes are presented since R^2 values for the other axes were less than 0.14 for all variables combined. All environmental variables measured as percentages were arc-sine transformed.

Dependent Variable = Ordination Axis 1

Number of variables in model	R^2	Independent Variables in Model		
1	0.00	% Organic		
1	0.01	Depth		
1	0.03	% Sand		
1	0.07	Sand-phi		
1	0.15	% $CaCO_3$		
1	0.21	Kilometer		
2	0.21	Kilometer	% Organic	
2	0.22	Kilometer	Depth	
2	0.22	Kilometer	% Sand	
2	0.22	Kilometer	Sand-phi	
2	0.22	% $CaCO_3$	Sand-phi	
2	0.31	Kilometer	% $CaCO_3$	
3	0.23	Kilometer	Sand-phi	% Organic
3	0.30	Kilometer	% Sand	% Organic
3	0.31	Kilometer	% $CaCO_3$	Depth
3	0.31	Kilometer	% $CaCO_3$	% Organic
3	0.31	Kilometer	% $CaCO_3$	% Sand
3	0.32	Kilometer	% $CaCO_3$	Sand-phi
6	0.34	All Variables		

Dependent Variable = Ordination Axis 2

Number of variables in model	R^2	Independent Variables in Model		
1	0.00	Sand-phi		
1	0.01	% $CaCO_3$		
1	0.03	% Organic		
1	0.03	% Sand		
1	0.06	Depth		
1	0.20	Kilometer		
2	0.15	Depth	% Sand	
2	0.20	Kilometer	% $CaCO_3$	
2	0.24	Kilometer	Depth	
2	0.24	Kilometer	Sand-phi	
2	0.24	Kilometer	% Organic	
2	0.28	Kilometer	% Sand	
3	0.27	Kilometer	Depth	Sand-phi
3	0.28	Kilometer	% Sand	% $CaCO_3$
3	0.28	Kilometer	% Sand	Sand-phi
3	0.28	Kilometer	% Sand	% Organic
3	0.30	Kilometer	% Organic	Depth
3	0.35	Kilometer	% Sand	Depth
6	0.36	All Variables		

Dependent Variable = Ordination Axis 3

Number of variables in model	R^2	Independent Variables in Model		
1	0.00	% $CaCO_3$		
1	0.00	Depth		
1	0.04	Kilometer		
1	0.05	% Sand		
1	0.06	% Organic		
1	0.16	Sand-phi		
2	0.09	% Organic	Kilometer	
2	0.17	Sand-phi	% $CaCO_3$	
2	0.17	Sand-phi	% Sand	
2	0.17	Sand-phi	Kilometer	
2	0.17	Sand-phi	Depth	
2	0.17	Sand-phi	% Organic	
3	0.17	Sand-phi	Depth	% $CaCO_3$
3	0.17	Sand-phi	Depth	Kilometer
3	0.17	Sand-phi	% Organic	% $CaCO_3$
3	0.17	Sand-phi	% Organic	Depth
3	0.18	Sand-phi	% Organic	Kilometer
3	0.19	Sand-phi	% Organic	% Sand
6	0.20	All Variables		

Table VII.19. Coefficients of separate determination indicating the importance of environmental variables on axes 1-3 of the discriminant analysis. Percent axes were arcsine transformed. The higher coefficients are underlined.

Environmental Variable	Discriminant Axis		
	Axis 1	Axis 2	Axis 3
Kilometer	<u>56.38</u>	4.95	0.86
Depth	4.59	6.47	6.63
Mean phi (sand)	11.11	<u>10.99</u>	<u>16.00</u>
% Sand	1.62	<u>56.82</u>	<u>25.05</u>
% $CaCO_3$	<u>25.47</u>	1.87	4.45
% Organic	0.84	<u>18.91</u>	<u>47.01</u>

sediments with more than 75% fine material (Appendix VI). The lack of a large gradient in sediment type among most of the stations, combined with the fact that many of the numerically dominant species were abundant over a range of sediment types probably contributed to the low correlation and discriminant coefficients observed. Finally, the lack of strong relationships between the ordination scores and the environmental variables, suggests that other parameters not measured (such as predation and competition) may be important in influencing the distribution and composition of benthic communities in this estuary.

Additional analyses were performed on subsets of the 178 station x 105 species matrix using the EAPS procedures described above in order to determine (1) whether further reductions of the data set would substantially improve the high variance explained in the ordination scores, and (2) whether partitioning the stations by area (i.e., harbor basin stations only, rivers stations only) would improve correlations with the sediment and hydrographic variables measured within those areas. The results from these analyses are not presented, however, since they did not significantly improve our interpretations of the data or alter the conclusions obtained from the larger data set with respect to the faunal distribution patterns.

Distribution of Benthic Assemblages in Relation to Anthropogenic Activities:

Our short-term spatially intensive study of the Charleston Harbor estuary did not reveal any clear relationships between the abundance or distribution of benthic organisms and the various human activities in the study area. The lack of any such patterns may be due partly to the considerable variance in the data set resulting from the large variety of habitats sampled, combined with the highly contagious distributions of many of the dominant species including *Mulinia lateralis, Paraprionospio pinnata, Heteromastus filiformis, Nereis succinea, Glycinde solitaria, Streblospio benedicti*, and others. Several of these species are known to be either pollution-tolerant or opportunistic organisms (Maurer *et al.*, 1974; Boesch, 1977b; Pearson and Rosenberg, 1978; Rhoades *et al.*, 1978; Holland, 1985; Holland *et al.*, 1989). While these species were not restricted to any particular station groups (Figure VII.30, Table VII.16), their high abundances at some sites may reflect anthropogenic effects. Additionally, as discussed below, a few sites showed evidence of reduced benthic diversity, low faunal abundance, or small scale differences in community composition that may be related to environmental perturbations.

Stations sampled within the harbor basin included a few sites adjacent to port facilities, as well as several sites in the maintained channels. Comparison of the species composition, diversity, and abundance of benthic organisms among these sites generally did not show any marked differences compared with stations in non-channel areas having similar sediment composition. Most of the harbor stations clustered into station groups A, C and D, with channel and non-channel sites represented in each of the station subgroups (Figure VII.31). A few of the samples collected from the channel reaches and anchorage basin near the harbor entrance, contained relatively few species and/or few organisms compared to other channel and non-channel sites in that portion of the estuary. This may reflect effects of a nearby dredging operation which was in progress during the sampling period. If so, the difference noted were limited in areal extent.

Only eight of the harbor basin samples clustered into station groups B and E (Figures VII.29, VII.31). Although the faunal composition at these sites was relatively dissimilar to the other stations sampled in this portion of the estuary, only three stations (CH12, CH65, CH66) had very few organisms or species. Station CH12 was adjacent to the dredging operation noted above. The sample from this station contained only four species, with nematodes representing 91% of the organisms. The samples from stations CH65 and CH66 were both depauperate ($\leq$8 organisms/sample) and contained very few species ($\leq$3 species/sample). These sites were close to the sewage outfall at Plum Island, which may have some influence on the benthic communities in that area. Although a much more rigorous sampling effort would be required to adequately evaluate the effects of sewage effluents on benthic communities in the area, it should be noted that the other stations near the Plum Island outfall did not show evidence of benthic disturbance compared to sites further from the outfall having similar sediment composition.

The other five harbor basin sites which clustered into station groups B and E included three channel stations and two stations near the harbor entrance outside the channel. Sediments at these stations were predominantly muddy sands. Although the cluster analysis indicated that the faunal composition at these sites differed from the other sites with muddy sand bottoms, faunal abundance, diversity and the types of benthic organisms present at these sites were not indicative of pollution stress (Appendix VII.B).

Among the three rivers sampled, the Cooper River has the greatest concentration of industrial and port facilities, all of which are located along the western shoreline (Figure VII.31). The east bank of the river is largely undeveloped, although large spoil areas are located there. Many of the channel reaches in this area are also periodically dredged to maintain naval and commercial port traffic. Benthic assemblages sampled at the 53 stations in this river, however, did not show any major differences in the community parameters measured that could be clearly related to these activities, especially when compared with similar but less developed areas in the Wando and Ashley River. For example, normal cluster analysis of macrofaunal samples collected in this river indicated that most of the stations located in or adjacent to the port facilities, channels, and industrial outfalls were associated with stations groups C and D (Figure VII.31). Many of these stations also contained high abundances of species such as *Paraprionospio pinnata, Heteromastus filiformis, Mulinia lateralis*, and oligochaetes (Figure VII.27), which are very common in southeastern estuaries and considered to be either opportunistic or pollution-tolerant species (Maurer, *et al.*, 1974; Boesch, 1977b; Pearson and Rosenberg, 1978). Thus, their high abundance at these sites may be indicative of bottom disturbance. On the other hand, station groups C and D contained many sites located in relatively undeveloped portions of the Cooper, Wando and Ashley Rivers and the same species were often very abundant at those sites as well. Stations near a major shipping terminal located in the lower portion of the Wando River, and stations near industrial activities in the Ashley River also showed no clear difference compared to less developed areas in these rivers.

As noted previously, species diversity values and faunal abundance showed considerable variability, with no clear pattern related to station location within each river system. Low values of H' and low numbers of species were detected at most sites located within or adjacent to the port facilities, industrial sites, and many of the channel locations, although similar values were observed at some of the stations in less developed areas of each river. Approximately 70% of the Cooper River stations had H' values of less than 2.0 bits/individual, with more than 76% of the stations having fewer than 10 species in the samples. In contrast, less than 48% of the stations in Wando River and 37% of the stations in the Ashley River had estimates of H' <2.0 bits/individual. Low diversities at many of the sites in all three river systems were often due to a low evenness value (J') resulting from numerical dominance by one or a few species. Several stations had very low faunal abundances as well, but these sites were often not close to any known source of anthropogenic disturbance.

In summary, the high degree of natural variability observed among the stations with respect to both the benthic and environmental variables tended to obscure evidence of biological stress that could be clearly related to various human activities in the harbor system. Although this survey was not designed to thoroughly evaluate the effects of each of these activities on the benthos, the low diversities and high abundances of certain opportunistic and pollution-tolerant species at several sites may be indicative of adverse effects from physical or chemical disturbance to the bottom communities. Additional studies are necessary in order to confirm whether some of the benthic distribution patterns noted are due to anthropogenic activities, particularly with regard to the effects from the Plum Island sewage outfall, and the port facilities. Based on the results of our survey, these studies should incorporate a sampling design that minimizes station variability related to salinity regimes and bottom sediment composition.

SUMMARY

1. Results from the four-year study of benthic macrofauna in the Charleston Harbor estuary indicate that the spatial distribution of these organisms is similar in many respects to that of other gradient estuaries along the mid-Atlantic, southeast, and Gulf coasts of the United States.

2. Cluster and nodal analyses showed clear distinctions among tidal freshwater, low salinity (oligo-mesohaline) and high salinity (meso-polyhaline) brackish water faunal groups. Within each salinity zone, benthic assemblages were further distinguished by their affinities for different sediment types.

3. Temporal patterns of distribution and abundance were not as readily apparent as spatial trends. Although total numbers of individuals and species varied greatly throughout the four-year study, there was no consistent seasonal or annual periodicity in these fluctuations.

4. Effects of rediversion on the benthos are difficult to infer from this study in the absence of a long-term pre-rediversion database; however, comparison of pre- and post-rediversion data collected in this study and other studies suggests that the qualitative composition of the macrobenthos has not changed markedly since rediversion.

5. Despite the lack of evidence for drastic alterations of the benthos, a few of the dominant species appeared to exhibit a trend toward decreasing abundance in certain reaches of the estuary.

6. Results from the more intensive short-term assessment of benthic macrofauna collected from 178 stations in the harbor basin and lower reaches of the three river systems identified community distribution patterns which were generally similar to those described in the four-year study.

7. Several environmental variables exhibited a weak correlation to the benthic community variables measured. These included station distance from the harbor entrance, salinity, percent sand, percent calcium carbonate, percent organic content of sediments, and sand grain size. Lack of a strong correlation between the physical and biological variables measured was due, in part, to a lack of well-defined gradients in the salinity and sediment regimes within the lower portion of the estuary.

8. Numerically dominant taxa included mollusks, polychaetes, oligochates, nematodes, and amphipods. These taxa were abundant throughout the study area, although their distribution was often very contagious.

9. Within the harbor basin, a few sites showed evidence of reduced benthic diversity, low faunal abundance, or small-scale differences in community composition that may have reflected perturbations from dredging operations or the sewage outfall at Plum Island.

10. Among the three river systems, average diversity values were lower in the Cooper River than in the Ashley and Wando Rivers. The Cooper River also had the highest percentage of stations with relatively low estimates of diversity ($H' < 2.0$).

11. The lower diversity in the Cooper River may reflect some adverse effects from the greater number of industrial and port facilities in this river compared to the other two river systems, however, the other community parameters did not show major

differences that could be clearly related to human activities. This may be due, in part, to the high degree of natural variability observed in both the biological and environmental variables, which would tend to obscure evidence of anthropogenic stress.

CHAPTER VIII:

FINFISH AND INVERTEBRATE COMMUNITIES

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Charleston, South Carolina

INTRODUCTION

The Charleston Harbor estuarine system supports a diverse assemblage of marine fish and crustaceans (Wenner *et al.*, 1984), and provides both a seasonal habitat for adults and juveniles of many species and a permanent, year-round habitat for residents. Many of the species present in the Harbor system are either commercially or recreationally valuable. For example, this estuary supports large populations of penaeid shrimps (*Penaeus setiferus, P. aztecus*, and *P. duorarum*) and blue crab (*Callinectes sapidus*) which are harvested both commercially and recreationally. The shrimp fishery is South Carolina's largest commercial fishery, averaging 3.24 million pounds (11.8 million dollars) annually during the past 10 years. The Charleston Harbor estuary contributes approximately 20% of the state's total shrimp landings (unpublished SCWMRD landings data for 1978 - 1987). Annual commercial landings of blue crab averaged 6.17 million pounds (1.7 million dollars) over the past 10 years, with Charleston Harbor accounting for about 8 % of the statewide total (unpublished SCWMRD landings data for 1978 - 1987).

Several species of fish in the estuary are important recreationally. Spot (*Leiostomus xanthurus*), Atlantic croaker (*Micropogonias undulatus*), red drum (*Sciaenops ocellata*), spotted seatrout (*Cynoscion nebulosus*), flounder (*Paralichthys lethostigma, P. dentatus*),and catfish (*Ictalurus catus, I. furcatus*) are all abundant and are consistently among the species most valued by recreational anglers (Low *et al.*, 1986; U.S. Department of Commerce, 1987). The economic value of marine sport fishing is difficult to assess, but estimates of total expenditures in South Carolina range from 40.3 million dollars in 1980 (U.S. Department of Interior, 1983) to 187 million dollars in 1983 (Low *et al.*, 1986).

In addition to the economically valuable species, the Charleston Harbor estuary also supports numerous species of ecological importance (Wenner *et al.*, 1984) such as the bay anchovy (*Anchoa mitchilli*) and the grass shrimps (*Palaemonetes pugio, P. vulgaris*) which serve as forage for many of the economically important species and are a vital part of the estuarine food web.

Concern over these valuable resources prompted several studies that provided data prior to rediversion. Wenner *et al.* (1984) reported on five years of monthly trawling conducted throughout the Cooper River and Charleston Harbor from 1973 through 1977 to determine biological and hydrographic conditions in the estuary. Shealy and Bishop (1979) looked at the biological communities and hydrography in the Cooper River under controlled low-flow conditions in an attempt to predict possible impacts of rediversion on this river system. Burrell and Carson (1979) provided a similar assessment in the Wando River during the same period of controlled low-flow. These studies, combined with earlier surveys of the Charleston Harbor area (Bears Bluff Laboratories, Inc., 1964; Sandifer *et al.*, 1980; Shealy *et al.*, 1974) provide data which characterize the harbor system prior to the Cooper River Rediversion Project. The purpose of the present study was to identify changes in species composition, abundance, and distribution of fish and invertebrates throughout the estuary in relation to the Cooper River Rediversion Project.

METHODS

Data Collection:

Eight stations located throughout Charleston Harbor (CH02), the Cooper River (CR01-CR04), and the Wando River (WR01-WR03) were sampled bimonthly over a four-year period beginning in November, 1984 (Figure VIII.1). An additional station in the lower harbor basin (CH03) was added in July, 1985 and three stations were added in the Ashley River (AR01-AR03) in September, 1987. Sampling at these stations (Figure VIII.1) continued bimonthly through November, 1988. Rediversion occurred during 27-31 August 1985. Sampling dates relative to rediversion are referred to later in this Chapter as follows:

Year 1	Nov., 1984 - July, 1985 (all pre-rediversion sampling)
Year 2	Sept., 1985 - July, 1986
Year 3	Sept., 1986 - July, 1987
Year 4	Sept., 1987 - July, 1988

Seasons were defined as:

Winter	January and March
Spring	May
Summer	July and September
Fall	November

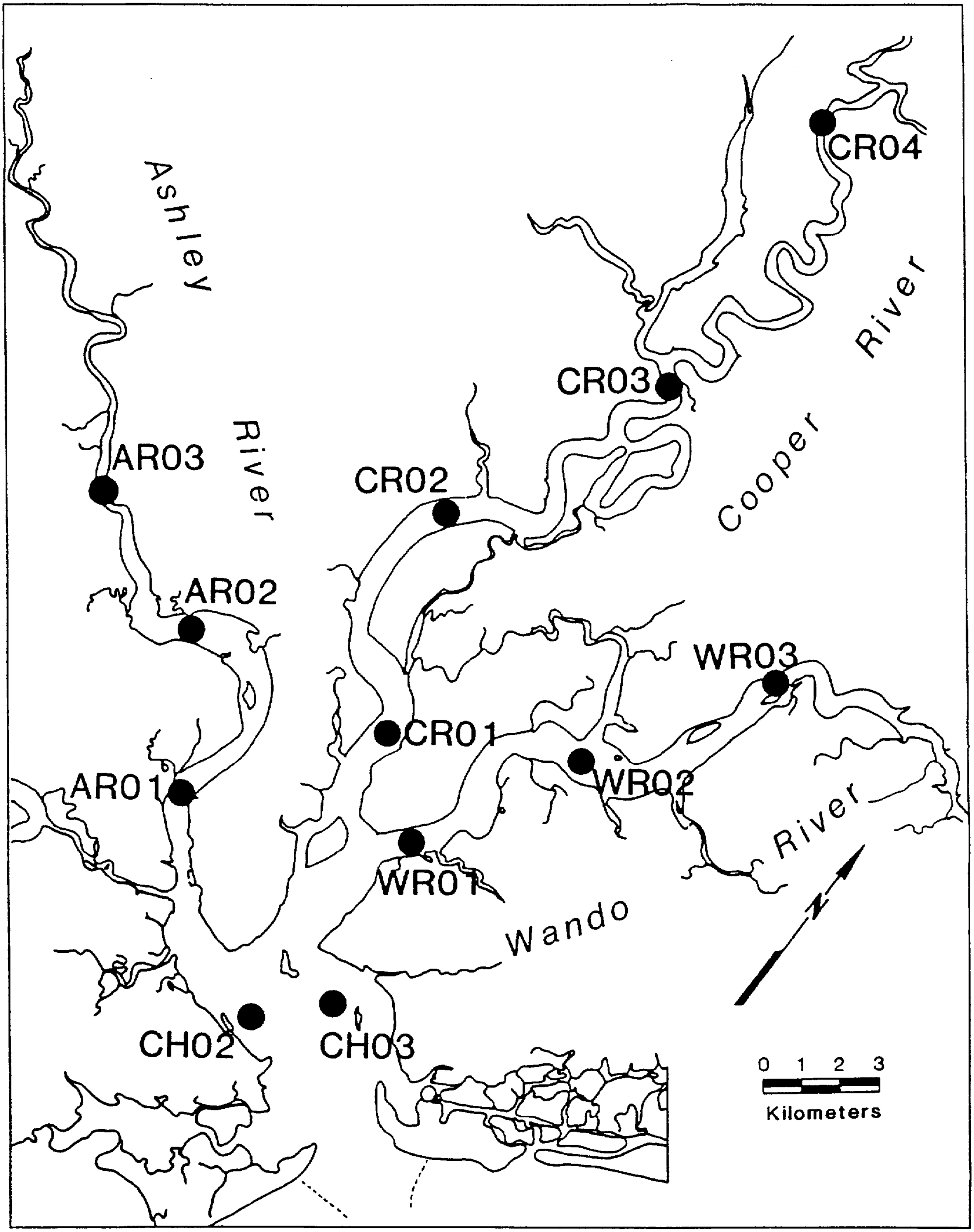


Figure VIII.1. Location of 12 trawl stations sampled throughout the Charleston Harbor estuary.

Three trawl tows were made at each station using a 7.5-m (25 ft) four-seam semi-balloon trawl with 38-mm (1.5 in) bar mesh equipped with a 4-mm (0.25 in) mesh liner in the cod end. Start and end positions were fixed to provide a standard tow distance of 0.75 km. The net was pulled in the direction of the prevailing tidal current. Trawling was restricted to within three hours of low tide during daylight hours in order to minimize sample variability due to tidal phase and diurnal changes. All trawling took place aboard the SCWMRD research vessel ANITA, a 16-m (52 ft) stern trawler. Surface and bottom hydrographic measurements (temperature, salinity, pH, and dissolved oxygen) were taken prior to trawling at each site using a Hydrolab Water Quality Data System (Model SVRZ-SU Surveyor II).

Specimens collected in each trawl were sorted and identified to species, counted, weighed, and measured (total or fork length for fishes, total length for shrimp, carapace width for crabs, mantle length for squid). After sorting, large collections of individual species were subsampled for length measurements typically using at least 35 animals. Species which could not be identified in the field were preserved in 10% Formalin and returned to the laboratory for identification.

Statistical Analyses:

To evaluate temporal and spatial variability using total catch, either an analysis of variance (ANOVA) with Duncan's multiple range test (Duncan's) or Kruskal-Wallis test (K-W) were performed on three standard parameters: number of individuals, weight, and number of taxa per tow. Actual values of these parameters did not meet the assumptions of the test, and $\ln(x+1)$ transformed values were used. When the data did not meet the assumptions of ANOVA as determined by a graphic test of normality and the F-max test for homogeneity of variance, only the results of the Kruskal-Wallis test were presented. In all cases significance was judged at the alpha = 0.05 level. Differences among stations in the three catch parameters were tested by season before and after rediversion. Biomass and density were estimated using the area swept method, which assumes an effective fishing width equal to 60% of headrope length (Wathne, 1959) being towed for a pre-measured distance of 0.75 km. Standard tows sampled approximately 3,400 m^2 of bottom.

The three previously mentioned statistical tests were also run to evaluate catch variables for each of 23 species. These species were selected because of their numerical abundance or specific commercial, recreational, or scientific interests. Only data from stations and months where the species occurred more than once were included. For testing yearly variability, data from stations in the Ashley River, the harbor station (CH03), and data collected in September were not included because these were sampled inconsistently during Year 1. Distributional plots in all figures included all data to give a total summary of observations. In all cases removal of inconsistently sampled data from the figures did not significantly affect the patterns illustrated.

Cluster and nodal analyses were used on data pooled by station and year to determine the similarity in sites (normal) and species (inverse). The Bray-Curtis similarity coefficient with log-transformed abundance values was used for cluster analysis with flexible sorting and a cluster intensity coefficient of beta = -0.25 (Clifford and Stephenson, 1975). Species occurring in less than 10% of the pooled categories were not included. Interrelationships of the site and species groups were interpreted using two-way coincidence tables resulting from nodal analysis (Boesch, 1977a), with constancy indicating the extent to which a species group occurs in all stations of a site group and fidelity indicating the degree to which a species group occurs only within that site group. Values for diversity (H'), evenness (J'), and species richness (S - 1 / ln N) were calculated and plotted for the pooled data set (Margalef, 1958; Pielou, 1975).

RESULTS AND DISCUSSION

Environmental Parameters:

Several trends were observed in hydrographic conditions during this survey (Table VIII.1). Since a complete analysis of hydrographic changes is discussed in Chapters III-V, only a brief discussion of hydrographic data taken concurrently with the trawls is included here. The highest mean flow rates and lowest mean salinities were encountered prior to rediversion, while mean temperatures were similar each year. Average flow rates compared before versus after rediversion by season differed most in winter and least in summer. Stations varied in mean salinities from 0.0 ppt at the uppermost station in the Cooper to 29.7 ppt in the Harbor. Most stations were characterized as polyhaline in the Venice system (Symposium on the Classification of Brackish Waters, 1958), although some stations in the Cooper and Ashley Rivers were in the mesohaline environment and the uppermost station in the Cooper River was in a limnetic environment.

Catch Analysis:

The 724 tows made during this study collected more than 800,800 individuals weighing 7,547 kg. These fish and invertebrates comprised 179 taxa in 73 families (Tables VIII.2 and VIII.3). Over 632,800 fishes weighing 5,524 kg were taken representing 124 species in 52 families. The ten numerically dominant species accounted for 95.4% by number and 66.0% by weight of all fishes taken (Table VIII.2). Decapod crustaceans and

Table VIII.1. Overall summary of collection and catch data from Nov. 1984 through Nov. 1988.

Grouping	$\bar{x}$ Temp. (°C)	$\bar{x}$ Sal. (°/oo)	$\bar{x}$ Flowrate (cfs)	# Indiv /tow	$\overline{wt}$ (kg) /tow	# Taxa /tow	Total Tows
Year 1	19.06	16.0	7695	657.6	14.3	14.5	116
Year 2	20.11	18.7	3855	858.8	11.0	17.6	161
Year 3	19.79	17.6	4736	922.5	8.4	17.1	161
Year 4	19.15	18.9	4286	1706.4	9.8	18.8	214
Pre-rediversion							
Winter	12.98	12.4	10058	853.5	21.5	12.8	42
Spring	23.03	18.7	6704	405.5	8.7	13.5	24
Summer	28.68	18.9	4879	772.2	11.4	19.2	27
Fall	15.68	16.7	6744	428.5	10.2	13.3	23
Post-rediversion							
Winter	10.78	17.9	4548	1272.8	8.9	15.4	177
Spring	21.75	18.1	4113	939.4	9.0	18.4	89
Summer	27.37	18.2	4094	1418.2	11.5	21.1	216
Fall	18.22	20.3	4273	861.8	8.2	16.1	126
All Years							
CH02	19.39	27.4		1075.2	14.8	22.3	74
CH03	19.98	29.7		895.0	11.4	26.4	63
CR01	19.74	24.2		1260.1	19.9	21.5	71
CR02	19.61	16.8		1201.8	8.5	19.7	75
CR03	19.66	5.6		1229.6	10.5	12.8	74
CR04	19.28	0.0		157.2	7.0	7.3	73
WR01	19.43	22.7		1680.5	10.8	20.8	75
WR02	19.68	20.5		1105.0	4.3	14.3	75
WR03	20.70	19.3		393.6	9.0	12.7	72
AR01	19.78	22.9		2527.2	15.4	23.7	24
AR02	19.62	17.5		2608.5	4.5	15.7	24
AR03	19.56	10.2		898.3	5.5	14.1	24
All	19.70	18.1	4958	1105.0	10.4	17.5	724

Table VIII.2. Total numbers and total weights of all fish species collected from 1984-1988.

	Number		Weight	
	Total	%	Total (kg)	%
Anchoa mitchilli	200292	31.65	228.520	4.14
Stellifer lanceolatus	135901	21.47	330.663	5.99
Leiostomus xanthurus	114640	18.12	1179.650	21.35
Micropogonias undulatus	51228	8.09	440.647	7.98
Cynoscion regalis	45035	7.12	324.248	5.87
Urophycis regia	19578	3.09	279.658	5.06
Bairdiella chrysoura	12655	2.00	332.723	6.02
Symphurus plagiusa	11400	1.80	96.222	1.74
Brevoortia tyrannus	7584	1.20	189.619	3.43
Ictalurus catus	5645	0.89	242.640	4.39
Menticirrhus americanus	2715	0.43	26.391	0.48
Trinectes maculatus	2631	0.42	15.096	0.27
Ictalurus furcatus	2564	0.41	649.357	11.75
Ariopsis felis	2539	0.40	268.523	4.86
Paralichthys lethostigma	2343	0.37	130.467	2.36
Anchoa hepsetus	1417	0.22	8.782	0.16
Etropus crossotus	997	0.16	11.525	0.21
Trichiurus lepturus	928	0.15	18.209	0.33
Prionotus tribulus	895	0.14	3.704	0.07
Chaetodipterus faber	877	0.14	16.544	0.30
Chloroscombrus chrysurus	872	0.14	2.918	0.05
Peprilus alepidotus	854	0.13	8.284	0.15
Ictalurus punctatus	808	0.13	22.339	0.40
Opsanus tau	659	0.10	50.913	0.92
Paralichthys dentatus	628	0.10	38.315	0.69
Centropristis philadelphica	548	0.09	26.982	0.49
Morone americana	483	0.08	42.877	0.78
Ophidion marginata	455	0.07	5.306	0.10
Scophthalmus aquosus	448	0.07	7.653	0.14
Selene vomer	420	0.07	2.694	0.05
Menidia beryllina	387	0.06	1.376	0.02
Bagre marinus	385	0.06	8.105	0.15
Ancylopsetta quadrocellata	301	0.05	4.286	0.08
Preprilus triacanthus	292	0.05	1.172	0.03
Urophycis floridana	286	0.05	12.263	0.22
Cynoscion nebulosus	259	0.04	11.909	0.22
Alosa aestivalis	197	0.03	0.875	0.02
Mugil cephalus	192	0.03	8.247	0.15
Synodus foetens	179	0.03	3.872	0.07
Lagodon rhomboides	172	0.03	3.421	0.06
Cynoscion nothus	165	0.03	1.150	0.02
Dasyatis sabina	156	0.02	109.694	1.99

Table VIII.2. Continued:

	Number		Weight	
	Total	%	Total (kg)	%
Alosa sapidissima	152	0.02	2.971	0.05
Hypsoblennius hentzi	150	0.02	1.207	0.02
Pomatomus saltatrix	134	0.02	23.470	0.42
Selene setapinnis	114	0.02	0.353	0.01
Dorosoma petenense	111	0.02	0.486	0.01
Centropristis striata	107	0.02	6.063	0.11
Prionotus scitulus	94	0.01	0.886	0.02
Ophichthus gomesi	90	0.01	3.723	0.07
Gobiesox strumosus	85	0.01	0.422	0.01
Citharichthys spilopterus	84	0.01	0.707	0.01
Rhizoprionodon terraenovae	51	0.01	10.661	0.19
Opisthonema oglinum	47	0.01	0.399	0.01
Larimus fasciatus	46	0.01	0.673	0.01
Syngnathus fuscus	37	0.01	0.096	<0.01
Stephanolepis hispidus	36	0.01	0.178	<0.01
Orthopristis chrysoptera	32	<0.01	1.767	0.03
Anguilla rostrata	28	<0.01	5.431	0.10
Ictalurus melas	28	<0.01	6.789	0.12
Brevoortia smithi	27	<0.01	9.271	0.17
Myrophis punctatus	26	<0.01	0.221	<0.01
Archosargus probatocephalus	24	<0.01	37.880	0.69
Syngnathus louisianae	22	<0.01	0.127	<0.01
Dorosoma cepedianum	20	<0.01	0.344	0.01
Mugil curema	16	<0.01	0.478	0.01
Morone saxatilis	15	<0.01	0.428	0.01
Scomberomorus maculatus	15	<0.01	0.183	<0.01
Caranx hippos	14	<0.01	0.429	0.01
Gobiosoma bosci	14	<0.01	0.014	<0.01
Prionotus evolans	13	<0.01	0.079	<0.01
Sphoeroides maculatus	13	<0.01	0.647	0.01
Astroscopus y-graecum	11	<0.01	0.642	0.01
Chilomycterus schoepfi	11	<0.01	0.517	0.01
Conger oceanicus	10	<0.01	0.979	0.02
Menidia menidia	10	<0.01	0.020	<0.01
Dasyatis sayi	8	<0.01	6.748	0.12
Rhinoptera bonasus	8	<0.01	30.350	0.55
Acipenser brevirostrum	8	<0.01	22.850	0.41
Sciaenops ocellata	8	<0.01	1.280	0.02
Sphyraena guachancho	7	<0.01	0.018	<0.01
Lepisosteus osseus	6	<0.01	11.450	0.21
Eucinostomus argenteus	6	<0.01	0.093	<0.01
Pogonias cromis	6	<0.01	72.588	1.31
Gobiosoma ginsburgi	6	<0.01	0.002	<0.01

Table VIII.2. Continued:

	Number		Weight	
	Total	%	Total (kg)	%
Syngnathus floridae	5	<0.01	0.047	<0.01
Lutjanus griseus	5	<0.01	0.068	<0.01
Acipenser oxyrhynchus	4	<0.01	7.000	0.13
Lagocephalus laevigatus	4	<0.01	0.089	<0.01
Syngnathus scovelli	4	<0.01	0.023	<0.01
Lutjanus synagris	4	<0.01	0.043	<0.01
Dasyatis americana	3	<0.01	5.520	0.10
Ogcocephalus radiatus	3	<0.01	0.100	<0.01
Rachycentron canadum	3	<0.01	0.383	0.01
Hypsoblennius ionthus	3	<0.01	0.015	<0.01
Gobionellus hastatus	3	<0.01	0.035	<0.01
Prionotus carolinus	3	<0.01	0.008	<0.01
Citharichthys macrops	3	<0.01	0.008	<0.01
Carcharhinus obscurus	2	<0.01	7.500	0.14
Mustelus canis	2	<0.01	2.000	0.04
Gymnura micrura	2	<0.01	0.717	0.01
Myliobatis freminvilli	2	<0.01	1.325	0.02
Mycteroperca microlepis	2	<0.01	0.099	<0.01
Eucinostomus harengulus	2	<0.01	0.007	<0.01
Stenotomus aculeatus	2	<0.01	0.011	<0.01
Mullus auratus	2	<0.01	0.003	<0.01
Chaetodon ocellatus	2	<0.01	0.017	<0.01
Gobionellus boleosoma	2	<0.01	0.003	<0.01
Ictalurus platycephalus	2	<0.01	0.275	<0.01
Cyprinus carpio	2	<0.01	12.000	0.22
Dasyatis centroura	1	<0.01	54.545	0.99
Alosa mediocris	1	<0.01	0.067	<0.01
Centropomus sp.	1	<0.01	0.145	<0.01
Urophycis earlli	1	<0.01	0.010	<0.01
Micropterus dolomieui	1	<0.01	0.001	<0.01
Membras martinica	1	<0.01	0.004	<0.01
Hippocampus erectus	1	<0.01	0.001	<0.01
Eucinostomus gula	1	<0.01	0.020	<0.01
Menticirrhus saxatilis	1	<0.01	0.085	<0.01
Chasmodes bosquianus	1	<0.01	0.004	<0.01
Hypleurochilus geminatus	1	<0.01	0.010	<0.01
Aluterus schoepfi	1	<0.01	0.001	<0.01
Ophichthus melanoporus	1	<0.01	0.010	<0.01
Megalops atlanticus	1	<0.01	0.001	<0.01
TOTAL	632,845	100.00%	5,524.499	100.00%

Table VIII.3. Total numbers and total weights of all invertebrate species collected from 1984-1988.

	Number		Weight	
	Total	%	Total (kg)	%
Penaeus setiferus	86930	51.98	834.655	41.26
Penaeus aztecus	18006	10.77	215.690	10.66
Lolliguncula brevis	13425	8.03	82.725	4.09
Trachypenaeus constrictus	12407	7.42	4.888	0.24
Palaemonetes vulgaris	7490	4.48	1.567	0.08
Callinectes sapidus	6156	3.68	660.469	32.65
Penaeus duorarum	3879	2.32	25.396	1.26
Acetes americanus	3695	2.21	0.057	<0.01
Callinectes similis	3413	2.04	68.996	3.41
Portunus gibbesii	2299	1.37	4.702	0.23
Periclimenes longicaudatus	2231	1.33	0.119	0.01
Rhithropanopeus harrisii	1383	0.83	0.556	0.03
Squilla empusa	1058	0.63	8.746	0.43
Panopeus herbstii	1006	0.60	2.001	0.10
Pagurus longicarpus	897	0.54	0.271	0.01
Libinia dubia	457	0.27	8.607	0.43
Palaemonetes pugio	370	0.22	0.147	0.01
Latreutes parvulus	311	0.19	0.020	<0.01
Neopanope sayi	279	0.17	0.250	0.01
Portunus spinimanus	248	0.15	3.317	0.16
Cancer irroratus	233	0.14	0.955	0.05
Leptochela serratorbita	154	0.09	0.013	<0.01
Ovalipes ocellatus	145	0.09	0.435	0.02
Eurypanopeus depressus	115	0.07	0.077	<0.01
Xanthidae	88	0.05	0.025	<0.01
Clibanarius vittatus	87	0.05	0.059	<0.01
Alpheus heterochaelis	78	0.05	0.121	0.01
Callinectes ornatus	78	0.05	1.477	0.07
Neopontonides beaufortensis	64	0.04	0.012	<0.01
Macrobrachium ohione	49	0.03	0.180	0.01
Libinia emarginata	34	0.02	3.630	0.18
Limulus polyphemus	31	0.02	90.000	4.45
Menippe mercenaria	28	0.02	2.349	0.12
Palaemonetes sp.	17	0.01	0.005	<0.01
Lysmata wurdemanni	16	0.01	0.015	<0.01
Pagurus pollicaris	12	0.01	0.065	<0.01
Alpheus armillatus	10	0.01	0.008	<0.01
Ogyrides alphaerostris	7	<0.01	0.002	<0.01
Upogebia affinis	7	<0.01	0.018	<0.01
Hepatus epheliticus	7	<0.01	0.179	0.01
Pleocyemata	7	<0.01	0.002	<0.01

Table VIII.3. Continued:

	Number		Weight	
	Total	%	Total (kg)	%
Alpheus normanni	6	<0.01	0.004	<0.01
Macrobrachium acanthurus	6	<0.01	0.048	<0.01
Hexapanopeus angustifrons	5	<0.01	0.004	<0.01
Brachyura	4	<0.01	0.001	<0.01
Squilla neglecta	4	<0.01	0.038	<0.01
Portunus sp.	2	<0.01	0.001	<0.01
Sicyonia dorsalis	1	<0.01	0.001	<0.01
Petrolisthes galathinus	1	<0.01	0.001	<0.01
Ovalipes stephensoni	1	<0.01	0.002	<0.01
Arenaeus cribrarius	1	<0.01	0.016	<0.01
Metoporhaphis calcarata	1	<0.01	0.001	<0.01
Pagurus politus	1	<0.01	0.001	<0.01
Alpheus sp.	1	<0.01	0.001	<0.01
Macrobrachium olfersii	1	<0.01	0.006	<0.01
TOTAL	167,242	100.00	2,022.931	100.00

selected invertebrates collected greater than 167,200 individuals and 2,022 kg and comprised 55 taxa in 21 families. Among these, the five most abundant species accounted for 82.7% by number and 56.3% by weight (Table VIII.3). These data indicated a diverse fauna of fishes and invertebrates similar to that reported elsewhere. Reid (1954) collected 122 species from 58 families over a one year period from Cedar Key, Florida, represented by over 13,000 individuals. Dahlberg and Odum (1970) found 31,637 specimens comprised of 70 species from 37 families by trawling for 14 months in Georgia. A four-year sampling period in North Inlet, SC, produced 31,419 individuals from 108 fish and invertebrate taxa (Ogburn *et al.*, 1988). Care must be taken, however, in making direct comparisons between studies because of major differences in sampling gears and techniques.

Several general trends were observed in the abundance, biomass, and number of species sampled (Table VIII.1 and Figure VIII.2). The mean number of individuals and taxa taken per tow were lowest in Year 1 while the mean weight was highest. In the years after rediversion there was a general increase in the mean number of individuals and taxa taken but a general decrease in the mean weight (Figure VIII.2). Similar trends were

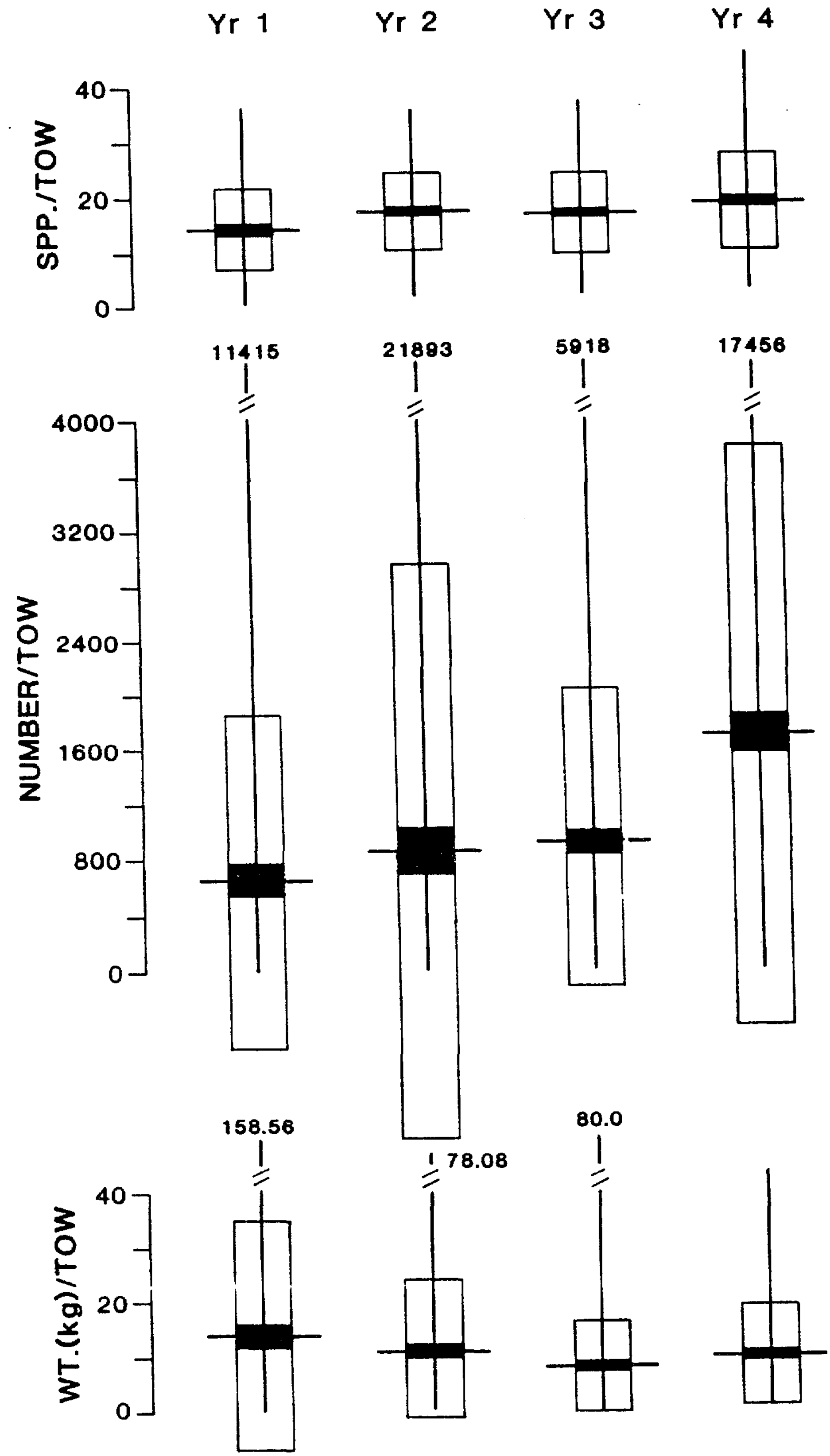


Figure VIII.2. Mean catch statistics by year for all stations combined. Vertical line represents range, open box represents one standard deviation, solid box represents standard error of the mean, and intersecting point represents mean.

observed between pre- and post-rediversion sampling in these three parameters (Table VIII.1). Prior to rediversion, the mean catch by number and weight was highest in winter, whereas summer catches were greatest after rediversion. More taxa per tow were taken consistently in summer throughout the survey. Typically, those stations closest to the mouths of the rivers had the highest mean catches by number, weight, and number of taxa, with these parameters generally decreasing upstream (Table VIII.1). Comparisons of the taxa collected in Year 1, the pre-rediversion year, with subsequent years showed a decrease in the percentage of these taxa reccurring, although there was an increase in the total number of taxa taken per tow (Figure VIII.3). This trend was also consistent for fish and invertebrate taxa considered separately.

Results of the statistical tests supported the general trends noted above, with significant differences observed among years in the log-transformed number of individuals and taxa per tow (Table VIII.4). Significantly more animals were collected in Year 4 and fewest were collected in Years 1 and 2. The number of taxa per tow was significantly lower in Year 1 compared to the three subsequent years. Similar results were found for fish and for invertebrates. No significant difference in log-transformed biomass captured per tow was found among the four years for all taxa combined, but yearly biomass differences were found for fishes and invertebrates when treated separately. Fish biomass was significantly greater in Year 1 than in Years 3 and 4. Invertebrate biomass was significantly greater in Year 4, with no significant differences among the remaining years.

Yearly differences in catch were also noted when log-transformed data were analyzed by station (Table VIII.4). Significant differences among years were found at all stations for number of individuals per tow. These differences were greatest between pre- and post-rediversion sampling at all stations except those in the lower portion of the estuary (CH02, CR01, WR01). The five upper stations were consistently higher with respect to the number per tow in the years following rediversion. In terms of biomass per tow, a significant difference was found among years at only three stations, but weight per tow for Years 1 and 4 were similar in each of these cases indicating yearly variability not resulting from rediversion. The log-transformed number of taxa per tow, while significantly different among years at five stations, was consistently higher after rediversion at only two stations, CR03 and WR03.

Further differences among stations were identified when log-transformed data collected before rediversion were tested by season (Table VIII.5). The number of individuals per tow was significantly higher at the lowest stations [WR01 (all seasons), CR01 (except summer), and CH02 (except summer)] and was significantly lower at the freshwater station (CR04) throughout the year. Catches at the lowest stations (WR01, CR01, and

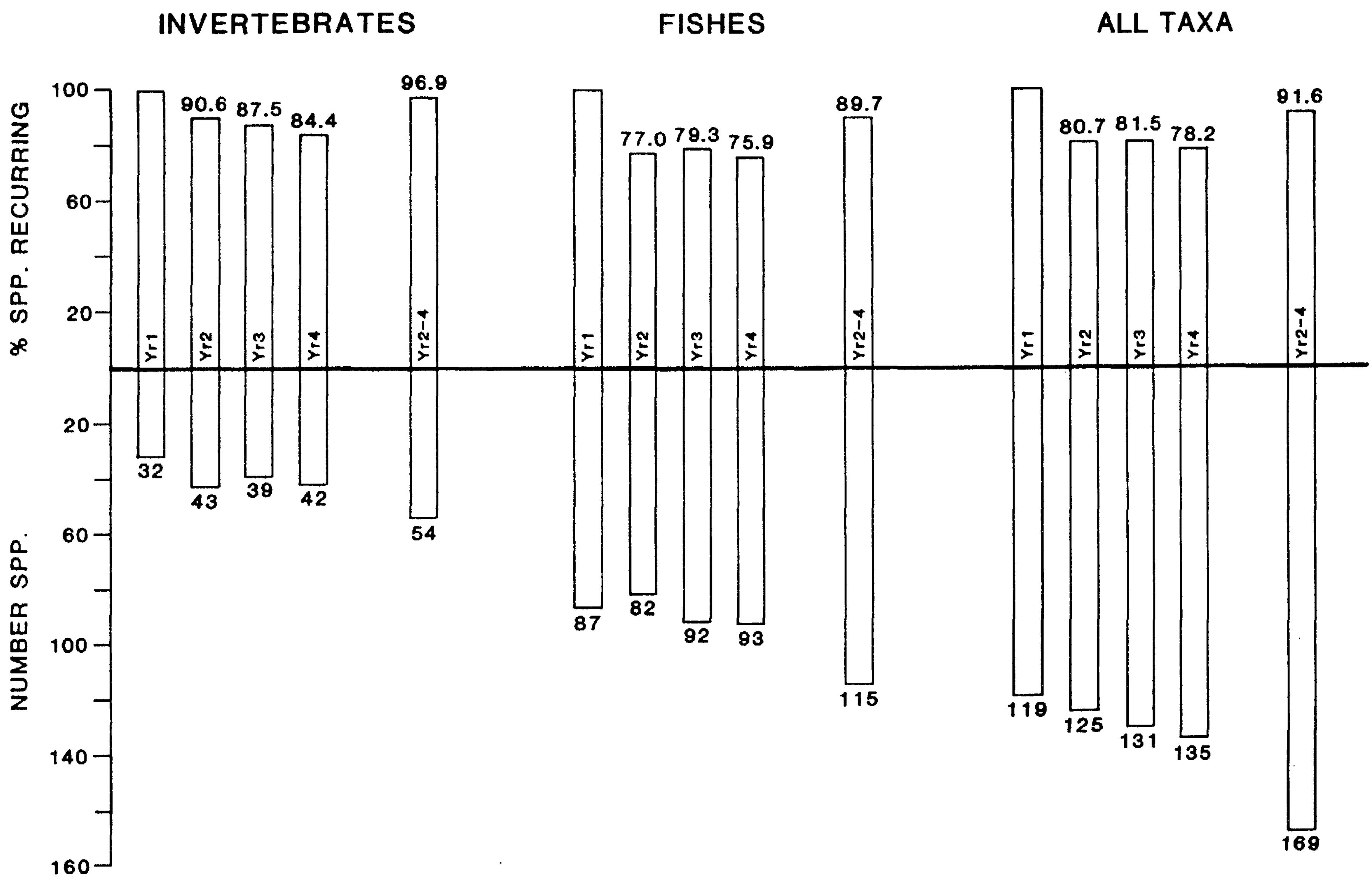


Figure VIII.3. Distribution by year of number of species taken at all stations combined and percentage of species from Year 1 recurring subsequently.

Table VIII.4. Results of statistical tests for differences among years by station based on log-transformed values of three parameters. Years underlined are not significantly different (ANOVA, Duncan's, α = 0.05 or K-W = Kruskal-Wallis, α = 0.05). ND = not significantly different; * = at $\alpha \leq 0.05$ level; ** = at $\alpha \leq 0.01$ level; *** = at $\alpha \leq 0.001$ level.

	# Individuals per Tow			Wt (kg) per Tow			# Taxa per Tow		
Station	Significant Difference	P	Year	Significant Difference	P	Year	Significant Difference	P	Year
CH02	*	0.03	4 1 3 2	*	0.04	1 4 3 2	*	0.04	4 2 3 1
CR01	*	0.02	4 2 1 3	***	0.0007	2 1 4 3	ND	0.39(K-W)	4 2 1 3
CR02	**	0.01	4 3 2 1	ND	0.38	4 2 1 3	**	0.004	2 4 3 1
CR03	***	0.0003	4 3 2 1	ND	0.08(K-W)	1 3 4 2	*	0.02	4 3 2 1
CR04	*	0.04	2 4 3 1	**	0.01	2 3 1 4	*	0.02	3 2 4 1
WR01	*	0.04	4 3 1 2	ND	0.09	1 4 2 3	ND	0.07	4 2 3 1
WR02	***	0.0001	3 4 2 1	ND	0.09(K-W)	4 3 1 2	ND	0.22	2 4 3 1
WR03	*	0.03	3 4 2 1	ND	0.22	3 1 2 4	**	0.01	4 3 2 1
Total of All Stations									
Invertebrates	***	0.0001	4 3 2 1	***	0.0001	4 3 1 2	***	0.0001	4 2 3 1
Fish	***	0.0001	4 3 2 1	*	0.05	1 2 3 4	**	0.003	4 2 3 1
All Taxa	***	0.0001	4 3 2 1	ND	0.23	1 2 4 3	***	0.0001	4 2 3 1

Table VIII.5. Results of statistical tests for differences among stations by season for sampling pre-rediversion (Nov. 1984 - July 1985). Stations underlined are not significantly different (ANOVA, Duncan's, α = 0.05 or K-W = Kruskal-Wallis, α = 0.05). ** = at $\alpha \leq 0.01$ level; *** = at $\alpha \leq 0.001$ level.

Parameter	Season	Significant Difference	P	Pre-Rediversion Stations								
# Individuals per tow (log-transformed)												
	Winter	***	0.0001	WR01	CRO2	CRO1	CHO2	CRO3	WRO3	CR04	WR02	
	Spring	***	0.001	CR01	CR02	CH02	CR03	WR02	WR01	CR04	WR03	
	Summer	**	0.007 (K-W)	WR03	WR01	CR03	CH02	WR02	CH03	CR01	CR02	CR04
	Fall	**	0.007 (K-W)	CH02	WR01	CR01	CR02	CR03	WR02	WR03	CR04	
wt (kg) per tow (log-transformed)												
	Winter	**	0.002	CR01	WR01	CR03	CH02	CR02	CR04	WR03	WR02	
	Spring	***	0.0001	CR01	CR02	CHO2	CR03	WR01	WR02	WR03	CR04	
	Summer	**	0.004	WR03	CH02	WR01	WR02	CH03	CR01	CR04	CR02	CR03
	Fall	***	0.0005	CR03	CH02	WR01	CR04	WR03	CR01	WR02	CR02	
# taxa per tow (log-transformed												
	Winter	***	0.0001	CR01	WR01	CH02	CR02	CR03	WR03	WR02	CR04	
	Spring	**	0.005 (K-W)	CR01	CR02	WR01	CH02	WR02	CR03	CR04	WR03	
	Summer	**	0.006 (K-W)	CH02	CH03	WR01	CR01	WR02	CR02	WR03	CR03	CR04
	Fall	***	0.0001	CH02	WR01	CR01	CR02	WR02	CR03	WR03	CR04	

CH02) contained significantly higher numbers of taxa, with fewest taxa being taken at the upriver stations (CR04, WR03, and CR03) during all seasons. Significantly higher weights per tow were taken at CH02 and CR01 (except fall) with lower weights typically at WR02 (except summer when CR03 lowest).

For the years after rediversion (Table VIII.6), tows at lower stations [WR01 (except winter), AR01, and AR02 (except spring)] contained significantly higher numbers of individuals per tow while lowest numbers occurred upriver [at CR04 and WR03]. The number of taxa per tow was highest at lower stations [CH03, AR01, CH02 (except spring), and CR01 (except spring)] and lowest upriver [at CR03, CR04, and WR03 (except spring)].

Table VIII.6. Results of statistical tests for differences among stations by season for sampling post-rediversion Sept. 1985 - Nov. 1988). Stations underlined are not significantly different (ANOVA, Duncan's, α = 0.05 or K-W = Kruskal-Wallis, α = 0.05). ** = at $\alpha \leq 0.01$ level; *** = at $\alpha \leq 0.001$ level.

Parameter	Season	Significant Difference	P	Post-Rediversion Stations
# Individuals per tow (log-transformed)				
	Winter	***	0.0001 (K-W)	AR01 AR02 CR02 CH02 CR01 WR01 CH03 AR03 CR03 WR02 WR03 CR04
	Spring	***	0.0001 (K-W)	CR03 WR01 AR03 AR01 CR02 CH03 CH02 WR02 AR02 CR01 CR04 WR03
	Summer	***	0.0001	AR01 WR01 AR02 CH02 CR01 CR03 CH03 CR02 WR02 AR03 WR03 CR04
	Fall	***	0.0001	AR01 AR02 CR03 WR01 CH03 CR02 WR02 AR03 CR01 CH02 WR03 CR04
wt (kg) per tow (log-transformed)				
	Winter	***	0.0001	AR01 CR01 CR02 WR01 CH03 CR03 WR03 CH02 AR03 CR04 AR02 WR02
	Spring	**	0.002	CH03 WR01 AR01 CH02 CR01 CR03 CR02 CR04 AR03 WR02 WR03 AR02
	Summer	***	0.0001	CH02 AR01 CR01 CH03 WR01 WR03 CR02 AR02 AR03 CR04 CR03 WR02
	Fall	***	0.0005	CR01 AR01 WR03 CH02 WR01 CH03 CR02 AR02 AR03 CR03 CR04 WR02
# taxa per tow (log-transformed				
	Winter	***	0.0001	CH03 CR01 CR02 CH02 WR01 AR01 CR03 WR03 AR03 WR02 AR02 CR04
	Spring	***	0.0001	AR01 CH03 WR01 CH02 WR02 CR01 CR02 AR03 WR03 AR02 CR03 CR04
	Summer	***	0.0001 (K-W)	AR01 CH03 CH02 CR01 WR01 CR02 AR02 AR03 WR02 CR03 WR03 CR04
	Fall	***	0.0001 (K-W)	CH03 CH02 AR01 CR01 CR02 WR01 WR02 AR02 AR03 CR03 WR03 CR04

Weight per tow was typically higher near mouths of rivers [AR01 and CR01 (except spring)] and lower elsewhere, especially at WR02.

The results of three previous studies in the Charleston Harbor system were compared to the present study (Bears Bluff Laboratories, Inc., 1964; Shealy and Bishop, 1979; Wenner *et al.*, 1984). Because of differing responses in avoidance and extrusion to varying sampling techniques, differences in towing regime must be considered carefully while making these comparisons. For example, in contrast to the earlier studies, trawl tows in this study were made with the tide instead of against it, a 4-mm liner was added in the cod end of the net, a larger trawl was used (7.5 m vs. 3 m for Bears Bluff and 6.1 m for the other two studies), and tows were made for fixed distances instead of fixed times. However, even considering these differences several patterns can be noted.

The Bears Bluff Laboratories, Inc., (1964) report on the Charleston Harbor estuary highlighted the importance of this system as a nursery area and indicated that the Ashley River yielded the highest abundances of fishes. These patterns were also noted in the present study following rediversion. Of the eight taxa previously listed as dominant, seven remained important in abundance.

The low-flow study by Shealy and Bishop (1979) expressed concern over possible changes in shrimp populations after rediversion. As discussed later, the present study documented a significant increase in the abundance and biomass of *Penaeus setiferus* and no significant changes in the abundance of *P. aztecus* other than yearly fluctuations, although an upriver shift was observed in the distribution of this species after rediversion. Additionally, other dominant species collected in their study were typical for the corresponding seasons in the present work.

The catches reported by Wenner *et al.* (1984) differed in some respects with the collections in this study. Ichthyological taxa that were previously dominant remained the dominant taxa; however, several differences were noted in the decapod crustaceans captured. In the earlier study, *Xiphopenaeus kroyeri* was the third most abundant decapod taxon, whereas none were taken in the present study. On the other hand, both *Acetes americanus* and *Periclimenes longicaudatus* were among the top invertebrate taxa in this study, but they rarely occurred in the earlier study. This may have been due, in part, to the addition of a liner in the net during the present study; however, a consistent significant increase in the number of taxa collected during the four years (including one pre-rediversion year) suggested that a greater number of species currently utilize the estuary. In the present study, there was a significant increase in number of fish per tow and decrease in the biomass per tow that may have been a result of smaller individuals utilizing this area after rediversion. Smaller size was indicated for ten of the 15 fish species selected for separate analyses, including three of the four dominant species. The decrease in fish weight combined with the concommittant increase in fish abundance observed in this study suggests that the Charleston Harbor estuary may be utilized more as a nursery ground by these species where formerly larger individuals were taken. An example of this was observed in the length frequency data for *Leiostomus xanthurus*. Both Wenner *et al.* (1984) and this study noted that spot >10 cm were frequently taken prior to rediversion. Subsequent to the time of rediversion, these larger animals did not occur in the catches.

Estimated mean densities at these stations for the years after rediversion ranged from 11,537 individuals per hectare at AR02 in winter to 152 in fall at CR04. Concentrations of animals were greatest in winter (3011/ha), intermediate in summer (2608/ha), and low in spring (1616/ha) and fall (1622/ha). Lowest densities were consistently at CR04 during all seasons. Mean biomass ranged from 46.9 kg/ha at CR01 to 3.7 kg/ha at AR02, with both extremes occurring in winter (Table VIII.7). Seasonal biomass estimates were highest in summer (20.1 kg/ha) but lower and similar for the rest of the year (16.1 kg/ha in winter, 15.2 kg/ha in spring, 14.3 kg/ha in fall).

In comparison with previous trawl studies, several differences in density of fishes and crustaceans were noted. The yearly density estimated after rediversion (2214 individuals/ha) was twice that reported previously from this area (1149/ha, Wenner *et al.*, 1984) and, in winter and spring, concentrations after rediversion were an order of magnitude greater than noted by Wenner *et al.* (1984). Fish and crustaceans in this estuarine habitat were also more concentrated than those in the shallow coastal sea off the southeastern United States (511/ha, Wenner and Sedberry, 1989), in Narragansett Bay (289/ha, calculated from Oviatt and Nixon, 1973), or in Long Island Sound (100/ha, calculated from Richards, 1963). However, densities were similar to those reported off Georgia (3850/ha, Hoese, 1973).

Mean yearly biomass was also higher after rediversion (16.4 kg/ha vs 11.0 kg/ha reported by Wenner *et al.*, 1984). Biomass in the estuary was similar (16.4 kg/ha) to that noted in offshore coastal waters (11.4 kg/ha, Hoese, 1973; 18.8 kg/ha, Wenner and Sedberry, 1989), and was higher than that reported from more polluted waters (7.6 kg/ha from Long Island Sound, calculated from Richards, 1963; 5-20 kg/ha from Boston, Mass., calculated from Haedrich and Haedrich, 1974; versus our 3.7-46.9 kg/ha). Our estimates were lower than biomass estimates reported from Narragansett Bay, R.I., however (31.9 kg/ha, calculated from Oviatt and Nixon, 1973).

Table VIII.7. Estimates of standardized catch by station for each season sampled after rediversion (Sept. 1985 - Nov. 1988).

Station	Mean #/ha				Mean kg/ha			
	W	Sp	Su	F	W	Sp	Su	F
CH02	1854	1365	2885	1259	12.3	19.1	42.1	16.6
CH03	1598	1317	1903	1433	13.8	28.8	25.7	14.2
CR01	1705	448	4601	1479	46.9	19.7	33.5	27.0
CR02	2851	1891	2393	1916	17.5	12.0	12.7	16.3
CR03	2297	4322	1977	2218	12.8	12.5	10.8	9.5
CR04	170	345	516	152	8.7	18.4	15.7	8.9
WR01	1722	2718	4761	1825	15.1	22.0	19.21	6.7
WR02	2865	2395	1856	2366	3.7	10.3	11.5	5.7
WR03	669	353	1061	3611	6.5	7.5	15.8	20.0
AR01	6675	1687	5186	26733	0.9	18.9	32.9	19.5
AR02	11537	733	2862	2436	4.3	4.2	11.1	9.2
AR03	2190	1817	1298	1343	11.2	8.6	10.3	8.1

Hoese (1973) concluded that the higher productivity of the marsh habitat was a major factor resulting in higher densities and biomass in a Georgia sound compared to offshore coastal waters. Data obtained from this study after rediversion support his observations for fish densities, but the data on biomass did not indicate greater productivity in the estuary. This could have resulted from differences in technique or timing of the studies. However, it is also possible that the offshore waters in this area were affected for a greater distance from shore by the estuaries, particularly considering the volume of discharge from the Santee-Cooper system.

Community Pattern Analysis:

Data analyses identified several major patterns in species catches. One pattern indicated no statistical difference in biomass and abundance among years (Table VIII.8).

Table VIII.8. Results of statistical tests for differences among years in number and weight for 23 selected species (Kruskal-Wallis, α = 0.05). N.D. = not significantly different; * = at $\alpha \leq 0.05$ level; ** = at $\alpha \leq 0.001$ level; *** = at $\alpha \leq 0.001$ level.

Species	Number per Tow			Weight per Tow		
	Significant Difference	P	Rank Order of Years	Significant Difference	P	Rank Order of years
Lolliguncula brevis	N.D.	0.25	2 1 4 3	N.D.	0.29	2 4 3 1
Penaeus aztecus	N.D.	0.17	4 1 2 3	N.D.	0.19	4 1 2 3
Penaeus duorarum	***	0.0001	2 1 3 4	*	0.02	2 1 3 4
Penaeus setiferus	***	0.0001	4 3 2 1	***	0.0001	4 3 2 1
Trachypenaeus constrictus	*	0.04	2 4 1 3	*	0.04	2 1 4 3
Palaemonetes vulgaris	*	0.03	4 3 2 1	*	0.05	3 4 2 1
Callinectes sapidus	***	0.0001	1 2 4 3	**	0.005	1 4 2 3
Calinectes similis	**	0.009	4 2 1 3	**	0.006	4 2 1 3
Brevoortia tyrannus	***	0.0002	3 1 2 4	***	0.0001	3 1 2 4
Anchoa mitchilli	***	0.0001	3 4 2 1	***	0.001	3 4 1 2
Ictalurus catus	N.D.	0.14	2 1 4 3	*	0.03	1 3 2 4
Ictalurus furcatus	N.D.	0.18	1 2 4 3	N.D.	0.09	1 2 3 4
Ariopsis felis	**	0.02	1 2 4 3	**	0.005	2 4 1 3
Urophycis regia	***	0.0001	4 3 1 2	***	0.0001	4 3 1 2
Bairdiella chrysoura	*	0.04	2 3 1 4	*	0.02	2 3 1 4
Cynoscion regalis	**	0.007	2 4 1 3	*	0.02	2 4 1 3
Leiostomus xanthurus	*	0.02	4 1 3 2	***	0.0001	1 2 4 3
Micropogonias undulatus	**	0.004	4 1 3 2	***	0.0001	1 2 4 3
Stellifer lanceolatus	*	0.04	2 1 3 4	N.D.	0.16	1 3 2 4
Paralichthys dentatus	N.D.	0.23	1 2 4 3	N.D.	0.07	1 2 4 3
Paralichthys lethostigma	**	0.01	2 3 4 1	*	0.02	4 2 3 1
Trinectes maculatus	*	0.03	3 1 2 4	*	0.03	1 3 4 2
Symphurus plagiusa	N.D.	0.77	1 4 2 3	N.D.	0.82	4 1 2 3

Among the 23 species considered, this occurred for five species - *Lolliguncula brevis* (brief squid, Appendix VIII.A.1), *Penaeus aztecus* (brown shrimp, Appendix VIII.A.2), *Ictalurus furcatus* (blue catfish, Appendix VIII.A.12), *Paralichthys dentatus* (summer flounder, Appendix VIII.A.20), and *Symphurus plagiusa* (blackcheek tonguefish, Appendix VIII.A.23). Two of these species, blue catfish and summer flounder, demonstrated marked changes in other aspects of their distribution and are discussed later. Changes in the distributions of the other three species were not observed during the study.

Eleven species had significant yearly fluctuations in abundance that did not appear to be an effect of rediversion (Table VIII.8). These were: *Penaeus duorarum* (pink shrimp, Appendix VIII.A.3), *Trachypenaeus constrictus* (roughneck shrimp, Appendix VIII.A.5), *Callinectes similis* (lesser blue crab, Appendix VIII.A.8), *Brevoortia tyrannus* (Atlantic menhaden, Appendix VIII.A.9), *Ictalurus catus* (white catfish, Appendix VIII.A.11), *Ariopsis felis* (sea catfish, Appendix VIII.A.13), *Urophycis regia* (spotted hake, Appendix VIII.A.14), *Birdiella chrysoura* (silver perch, Appendix VIII.A.15), *Cynoscion regalis* (weakfish, Appendix VIII.A.16), *Stellifer lanceolatus* (star drum, Appendix VIII.A.19), and *Trinectes maculatus* (hogchoker, Appendix VIII.A.22). An example of these fluctuations was shown in Figure VIII.4. All but star drum also showed significant fluctuations in biomass per tow. Significant yearly fluctuations in several species, *i.e.*, pink shrimp and Atlantic menhaden, are well documented (Nelson *et al.*, 1977; South Atlantic Fisheries Management Council [SAFMC], 1981). High commercial interest in these two species have resulted in the development of models to predict these yearly fluctuations. Wenner *et al.* (1984) noted yearly fluctuations in abundance of Atlantic menhaden, silver perch, star drum, and hogchoker, but observed consistent abundance of weakfish during pre-rediversion sampling in the Charleston Harbor estuarine system. Similar fluctuations have been noted in pink shrimp, roughneck shrimp, and spotted hake in other South Carolina estuaries (Ogburn *et al.*, 1988).

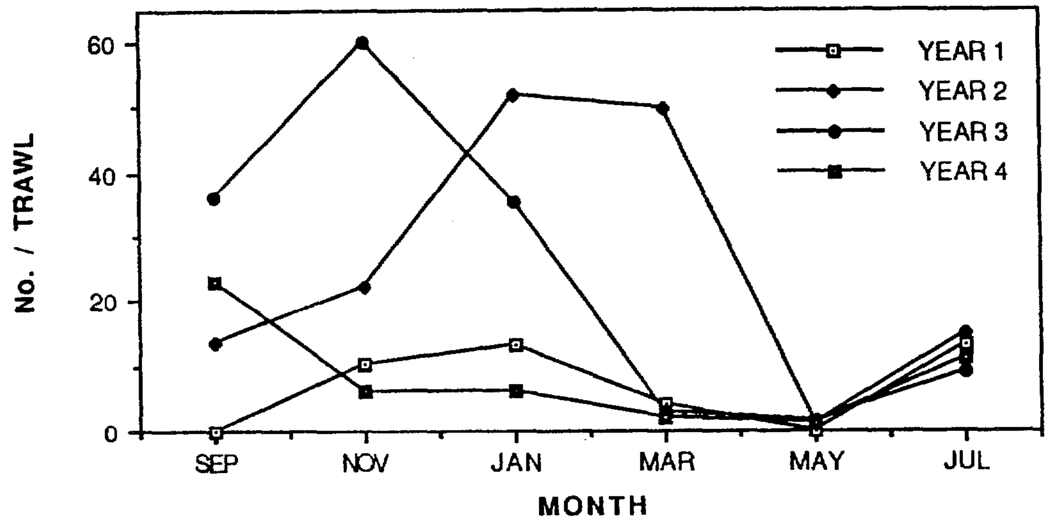
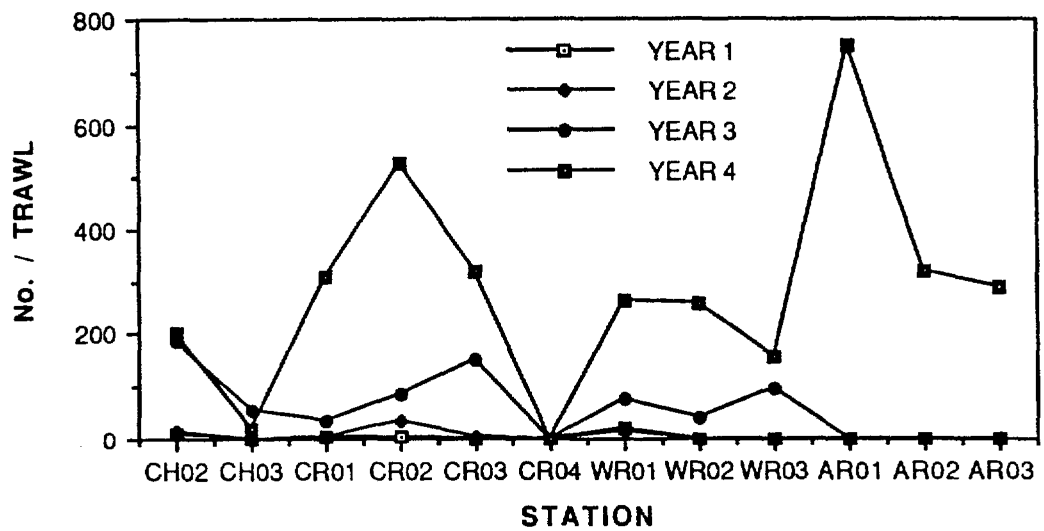


Figure VIII.4. Mean abundance of *Bairdiella chrysoura* captured per tow by month during the four-year study.

Significant increases in abundance and biomass after rediversion occurred for *Penaeus setiferus* (white shrimp, Appendix VIII.A.4), *Palaemonetes vulgaris* (grass shrimp, Appendix VIII.A.6), *Anchoa mitchilli* (bay anchovy, Appendix VIII.A.10), and *Paralichthys lethostigma* (southern flounder, Appendix VIII.A.21). Some of these changes may reflect fluctuations that are more complex than simply an effect of rediversion. For example, the population of white shrimp in South Carolina has been shown to be related to the duration of cold temperature ($\leq 6°$ C), the lowest temperature, and winter salinities (Whitaker, 1984). These winter conditions have direct impact on the spring landings. Fall landings from the central coast of South Carolina have been shown to vary inversely with salinity values from Charleston Harbor in August, provided a certain minimum recruitment survived during the spring (Lam *et al.*, 1989). This relationship was used to predict landings of shrimp. Calculated and observed landings data indicated that 1984 and 1985 were the lowest shrimp landings since 1977 because of the sustained low winter temperatures (Lam *et al.*, 1989). Subsequently, white shrimp catches in South Carolina have increased since 1985 [unpublished SCWMRD landings data for 1988)]. Hence, the significant increase in white shrimp catches after rediversion may reflect recovery from stressful winter conditions that occurred in 1984 and 1985. If salinities had increased, particularly in summer, resulting from decreased flow after rediversion, catch of white shrimp would have decreased. However, we noted a slight decrease in mean summer salinities (Table VIII.1) and catches significantly increased in both number and biomass (Figure VIII.5). The decrease in



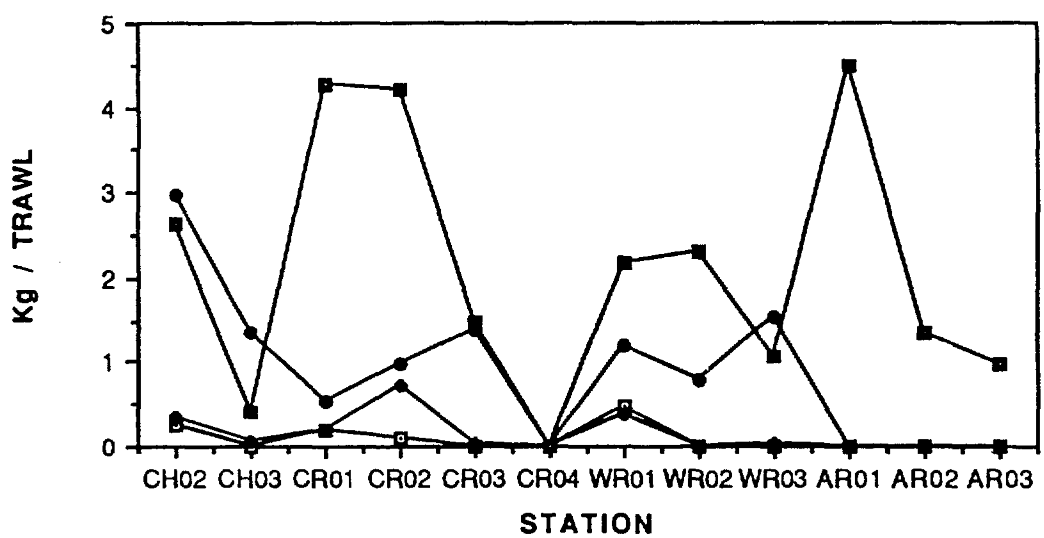


Figure VIII.5. Mean abundance and biomass of *Penaeus setiferus* captured per tow by salinities may reflect the effects of an excessive drought that persisted earlier in the study and a subsequent normalization of conditions. For the other three species exhibiting this pattern (grass shrimp, bay anchovy, and southern flounder), commercial landings exist only for southern flounder where catches are insignificant and tend to reflect effort and market changes rather than fluctuations in populations. Catches of these species during the current study reflected changes similar to those noted for white shrimp, indicating significant increases in abundance and biomass.

In contrast to the patterns noted for shrimp, a decrease in both abundance and biomass was noted for *Callinectes sapidus* after rediversion (Figure VIII.6, Appendix VIII.A.7). Previous sampling from 1973-1977 had indicated increasing abundances over the study period with a slight reduction in 1977 (Wenner *et al.*, 1984). Given the wide tolerance

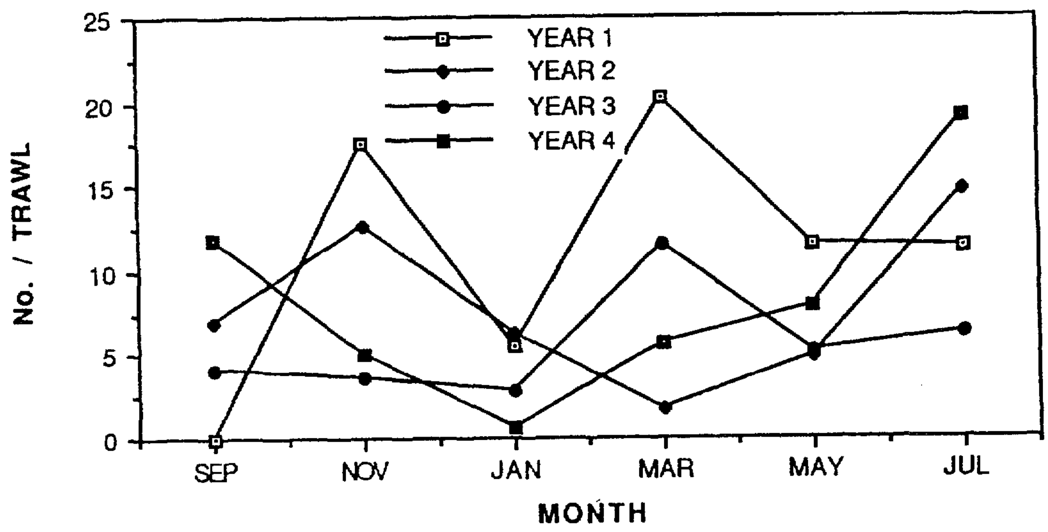


Figure VIII.6. Mean abundance of *Callinectes sapidus* captured per tow by month during the four-year study.

for salinity fluctuations in larval, juvenile, and adult blue crabs (Van Den Avyle and Fowler, 1983), it was not certain that the increased salinities resulting from rediversion could be affecting the observed decreases. In fact, it may have been expected that the increased salinities would have resulted in increases in abundance and biomass. There was an increase in landings of blue crabs from the Charleston area and from the entire state after rediversion (unpublished SCWMRD landings data, 1989). However, landings data may have frequently indicated influences of socio-economic factors rather than population fluctuations. Catches reported herein were taken from trawls in the main channel and may have underestimated changes in other sub-habitats of blue crabs. Catch data from the trawls appeared to vary inversely with the landings of blue crabs from this area. It was possible that decreased trawl catches reflected increased removal by crabbers, but data to evaluate this was not taken within each river.

Two species, *Leiostomus xanthurus* (spot, Appendix VIII.A.17) and *Micropogonias undulatus* (Atlantic croaker, Appendix VIII.A.18) showed significant increases in abundance and decreases in biomass after rediversion (Table VIII.8, Figure VIII.7). This could have resulted from increased catches of young utilizing the estuary. One possible explanation of why this occurred was that the apparent increase in young was the result of using a liner

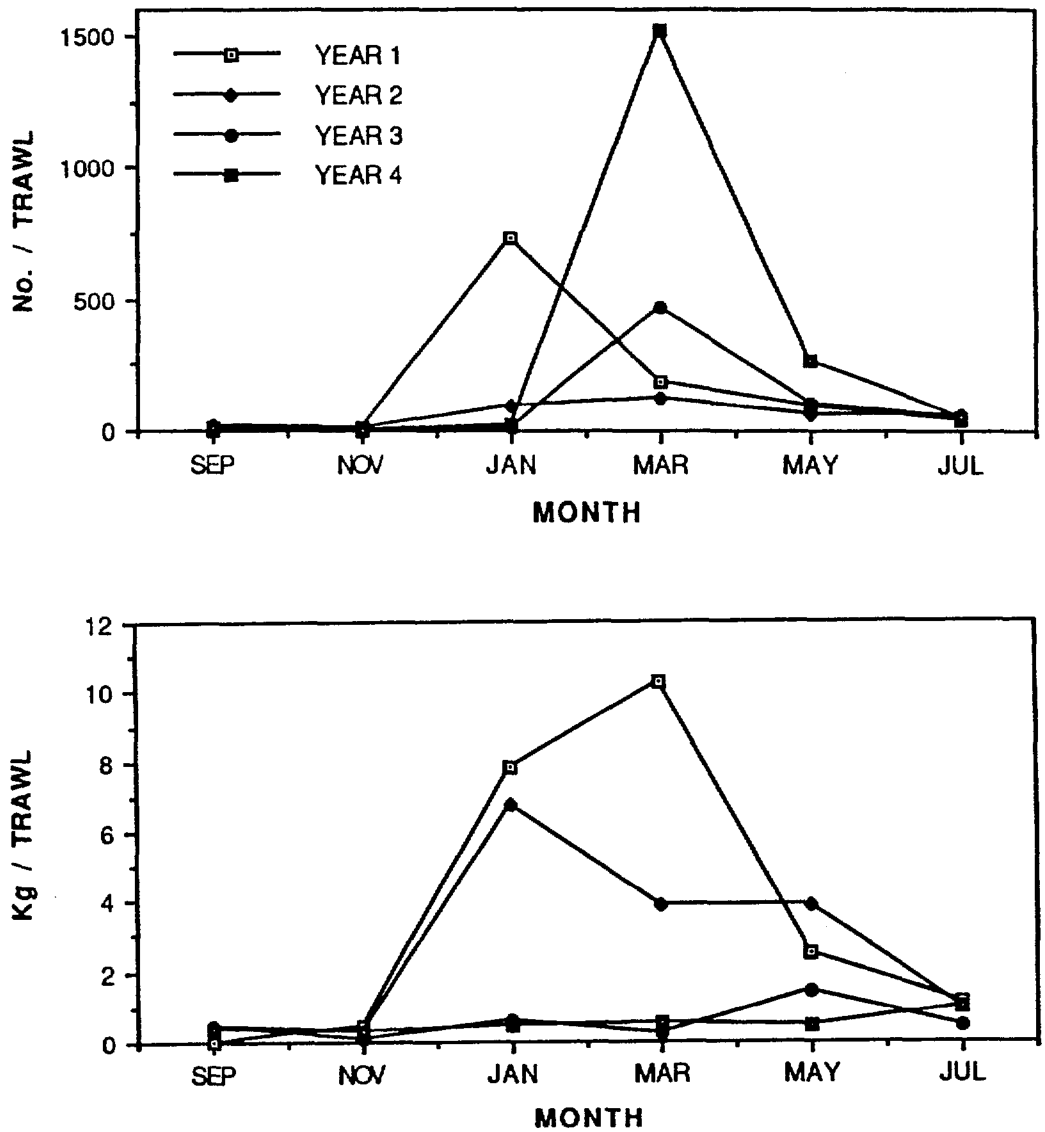


Figure VIII.7. Mean abundance and biomass of *Leiostomus xanthurus* captured per tow by month during the four-year study.

in the trawl during this study and, therefore, catching smaller individuals. However, both in Year 1 and in previous studies which did not use a net liner (Shealy *et al.*, 1974; Wenner *et al.*, 1984), the spot collected were primarily >7 cm (over 90% of catch in Year 1). Given this trend and the fact that specimens <5 cm composed over 90% of the catch after rediversion, it was not possible that catch biases resulting from the use of a liner explained the results. The similarity in sizes of spot and croaker during Year 1 and previous studies further minimizes biases resulting from differences in sampling technique as an explanation of the observed changes. Factors external to changes in the Charleston Harbor system may have contributed to recruitment being reduced in Year 2 and exceptional in Year 4. Landings data on spot and croaker were too insignificant to use in evaluating changes in the populations. Both species were recruited into the estuary primarily in the winter and spring. Winter was the season during which the greatest decreases in flowrate and increases in salinity occurred. However, both of these changes were expected to cause decreases in abundances of young. Weinstein *et al.* (1980) showed that young spot utilized vertical migration to effect a net upstream transport during flooding tides to reach fresh water nursery areas. Decreases in river flowrates would have reduced upriver velocities along the bottom hampering upstream transport. While the increased salinities were not expected to affect the euryhaline young, these conditions would result in greater distances for the spot to travel to reach their nursery grounds thereby increasing obstacles to survival and increased abundances. However, reductions in flowrates to a low and stable constant produced an estuary with fewer environmental fluctuations. These stable conditions may have promoted larval development and growth under less taxing conditions thereby increasing survival. Dominant species such as spot and croaker could have exploited these conditions.

Eight additional species showed marked increases in the percentage of smaller individuals over the four-year study (Figure VIII.8). These were *Penaeus setiferus* (Appendix VIII.A.4), *Anchoa mitchilli* (Appendix VIII.A.10), *Ictalurus catus* (Appendix VIII.A.11), *Ictalurus furcatus* (Appendix VIII.A.12), *Cynoscion regalis* (Appendix VIII.A.16), *Stellifer lanceolatus* (Appendix VIII.A.19), *Paralichthys dentatus* (Appendix VIII.A.20), and *Paralichthys lethostigma* (Appendix VIII.A.21). Two of these, *I. catus* and *I. furcatus*, were freshwater species that experienced a loss of available habitat within the surveyed area as salinities increased after rediversion. Reduction in their sizes possibly represented a reduction in the occurrence of adults and continued occurrence of the young having a greater tolerance to salinity fluctuations than the adults. The other six species were typical estuarine species whose young may be utilizing a more stable habitat similar to spot and croaker.

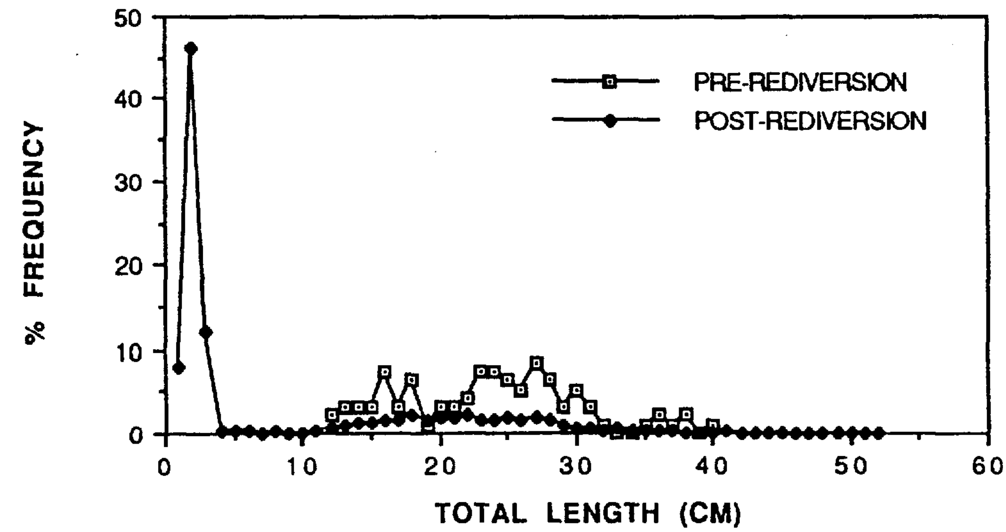


Figure VIII.8. Length-frequency distribution for catch of *Paralichthys lethostigma* for all stations sampled over the four-year period.

An upriver shift of 11-22 km (6-12 mi) in peak abundance occurred for nine species. The animals occurring further upriver after rediversion were *Penaeus duorarum* (Appendix VIII.A.3), *Penaeus setiferus* (Appendix VIII.A.4), *Callinectes sapidus* (Appendix VIII.A.7), *Anchoa mitchilli* (Appendix VIII.A.10), *Ictalurus catus* (Appendix VIII.A.11), *Ictalurus furcatus* (Appendix VIII.A.12), *Leiostomus xanthurus* (Appendix VIII.A.17), *Micropogonias undulatus* (Appendix VIII.A.18), and *Trinectes maculatus* (Appendix VIII.A.22). Most of these changes occurred in the Cooper River between either the lower two or upper two stations (Figure VIII.9). Upriver shifts in the Cooper River reflected the shifting salinity

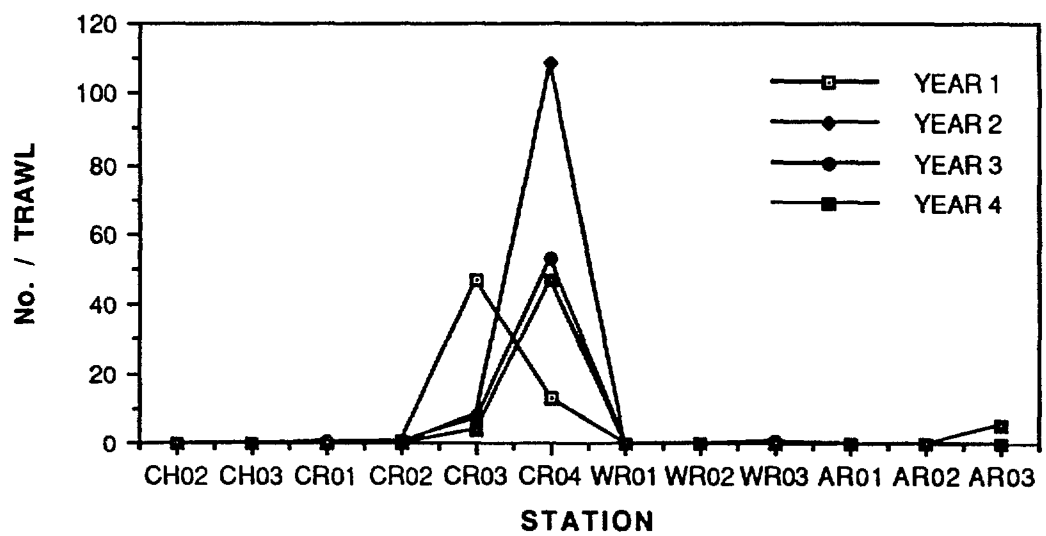


Figure VIII.9. Mean abundance of *Ictalurus catus* captured per tow by station during the four-year study.

regimes that resulted from altered flowrates. Two species, *Penaeus duorarum* and *Anchoa mitchilli*, had shifts only in the Wando River while two others, *Penaeus setiferus* and *Leiostomus xanthurus*, had shifts in both the Wando and Cooper Rivers. Prior to rediversion, salinities < 15 ppt were encountered at the lowest Wando River station, while, subsequently, this lower salinity water was only rarely encountered at the upper most Wando station. However, most of these species were euryhaline and shifts may have been the result of some secondary change, *e.g.*, distribution of prey organisms, that may have responded to the salinity change.

Species Observations:

In an effort to investigate more specific changes over the four-year study, observations on the twenty-three species were considered. Comments on yearly variation in abundance and biomass were given in the pattern analysis section and not repeated here. Families are presented phylogenetically.

Loliginidae - *Lolliguncula brevis* (brief squid) were present in the estuary during all seasons except winter, with abundance generally highest in spring and summer (Appendix VIII.A.1). This is consistent with Ogburn *et al.* (1988) and agreed with other findings which indicated that *L. brevis* migrate into estuaries in the spring and emigrate in fall during declining water temperatures (Laughlin and Livingston, 1982). Brief squid were limited in distribution to high salinity stations. They occurred throughout the Wando River, but stations CR02 and AR02 usually represented the upper limits for this species in the Cooper and Ashley, respectively. Tests on the salinity tolerance of this squid have shown severe osmotic stress in salinities below 17 ppt. (Hendrix *et al.*, 1981). This corresponds very closely with mean salinities at CR02 and AR02 (Table VIII.1).

Penaeidae - Shrimp of this family had a wide distribution throughout the Charleston Harbor estuary, but abundances varied according to species. *Penaeus aztecus* (brown shrimp) were abundant only during July (Appendix VIII.A.2). This seasonal occurrence was consistent with previous studies in South Carolina (Bishop and Shealy, 1977; Ogburn *et al.*, 1988; Wenner *et al.*, 1984). Highest abundances were in the harbor, at station CH02. Length-frequencies were similar each year, with almost 80% of all brown shrimp measuring 10 cm or more in total length.

Pink shrimp (*Penaeus duorarum*, Appendix VIII.A.3) were the least abundant of the penaeids. Historically, they have been less common in Charleston Harbor than the other species. It has been theorized that *P. duorarum*, being the most halophyllic of the penaeids, prefer higher salinity estuaries (Gunter *et al.*, 1964; Bishop and Shealy, 1977). Their distribution shifted after rediversion, moving upriver as the salinity gradient changed.

White shrimp (*Penaeus setiferus*) were the most abundant penaeid shrimp and the most numerous invertebrate collected throughout the study (Table VIII.3). Catches were seasonal, with most white shrimp caught from late summer through early winter (Appendix VIII.A.4). This was consistent with Ogburn *et al.* (1988) and Wenner *et al.* (1984). White shrimp were widely distributed throughout the estuary and this distribution increased after rediversion. Prior to rediversion, they were collected as far upriver as stations CR03 and WR02. After rediversion, they were found at all stations, although peak abundances remained at lower river stations.

Roughneck shrimp (*Trachypenaeus constrictus*, Appendix VIII.A.5) were most abundant in the summer, with smaller catches continuing year-round, similar to the seasonal pattern reported by Ogburn *et al.* (1988). They were most abundant at high salinity stations but this distribution was unaffected by rediversion.

Palaemonidae - The grass shrimp *Palaemonetes vulgaris* was collected primarily in summer before rediversion and in both late winter and summer after rediversion (Appendix VIII.A.6). They occurred throughout the estuary but were typically most numerous at the upriver station in the Wando (WR03).

Portunidae - The swimming crabs *Callinectes sapidus* (blue crab, Appendix VIII.A.7) and *C. similis* (lesser blue crab, Appendix VIII.A.8) had different seasonal catch patterns. *Callinectes sapidus* were collected year-round and at every station in the estuary but were most abundant at higher salinity stations. Other studies in this area also reported year-round occurrences (Archambault *et al.*, 1990; Ogburn *et al.*, 1988; Wenner *et al.*, 1984). The lesser blue crab was much more seasonal and was caught in abundance only from July to November. This was consistent with the findings of Ogburn *et al.* (1988) in the North Inlet estuary and of Tagatz (1967) in the St. Johns River estuary in Florida. They were caught primarily at the high salinity stations, although after rediversion they were taken occasionaly as far up the estuary as CR02 and CR03.

Clupeidae - *Brevoortia tyrannus* (Atlantic menhaden) were collected year-round with peak abundance during March (Appendix VIII.A.9). Ogburn *et al.* (1988) also found peak abundance of menhaden during late winter-early spring in North Inlet, S.C., although Wenner *et al.* (1984) found little change in abundance by month in the Cooper River and Charleston Harbor. Atlantic menhaden were also collected at every station sampled.

Engraulidae - *Anchoa mitchilli* (Bay anchovy) were the most abundant fish collected during the study, accounting for 31.6% of the total number of fishes caught. They were collected every month with no distinct pattern of seasonality (Appendix VIII.A.10), which is consistent with pre-rediversion sampling (Wenner *et al.*, 1984) as well as with studies conducted in other estuaries throughout South Carolina (Shealy *et al.*, 1974) and Georgia (Dahlberg and Odum, 1970). Ogburn *et al.* (1988), however, found greatest numbers in summer collections in North Inlet, S.C. Bay anchovy were collected at every station, although rarely at CR04, a primarily freshwater station. The greatest abundance of anchovies was at the mid-river stations, AR02, CR02, and WR02.

Ictaluridae - *Ictalurus catus* (White catfish) and *I. furcatus* (blue catfish) were captured year-round, with white catfish declining in numbers in November and January and blue catfish declining in January and July (Appendix VIII.A.11, VI.A.12). Both species were caught primarily at the upper stations in the Cooper River (CR03, CR04).

Ariidae - *Ariopsis felis* (sea catfish) were collected primarily during the warmer months only, May through September (Appendix VIII.A.13). This did not change with rediversion and remains consistent with earlier trawling surveys here and elsewhere (Ogburn *et al.*, 1988; Shealy *et al.*, 1974).

Gadidae - *Urophycis regia* (spotted hake) were highly seasonal in abundance, generally taken only in March and May (Appendix VIII.A.14). This corresponded to previous surveys of the Charleston Harbor estuary, as well as other estuaries in South Carolina (Ogburn *et al.*, 1988; Shealy *et al.*, 1974; Wenner *et al.*, 1984) and Georgia (Dahlberg and Odum, 1970). During the remainder of the year, these fish are known to migrate offshore (Hildebrand and Cable, 1938). They were collected primarily at the more seaward stations, either directly in the harbor (CH02, CH03) or at the lowest river stations (AR01, CR01, WR01).

Sciaenidae - Five sciaenid fishes were among the numerically dominant species collected. These were *Bairdiella chrysoura* (silver perch), *Cynoscion regalis* (weakfish), *Leiostomus xanthurus* (spot), *Micropogonias undulatus* (Atlantic croaker), and *Stellifer lanceolatus* (star drum). Seasonal distribution of most of these fishes (Appendix VIII.A.15-19) agreed with generally accepted abundance patterns, with silver perch declining in numbers during spring and early summer, weakfish abundant only during summer, and star drum abundant during summer and fall (Dahlberg and Odum, 1970; Ogburn *et al.*, 1988; Shealy *et al.*, 1974; Wenner *et al.*, 1984).

Spot was the third most abundant fish species collected and, although present year-round, were most abundant during winter and spring when young-of-the-year entered the estuary. Prior to rediversion, peak abundances occurred in January, while for the three years after rediversion it occurred in March (Appendix VIII.A.17). Spot are known to spawn off the continental shelf from November to March, with young-of-the-year entering southeastern estuaries from January through April (Dawson, 1958; Kobylinski and Sheridan, 1979; Warlen and Chester, 1985). Shealy *et al.* (1974) and Wenner *et al.* (1984) found ingress of juvenile spot during April in the Charleston Harbor estuary while Ogburn *et al.* (1988) found juveniles entering North Inlet during March. Spot were collected at every station throughout the estuary.

Atlantic croaker were also collected year-round, with greatest abundances occurring from March through July (Appendix VIII.A.18). Wenner *et al.* (1984) found greatest numbers from April through July. Length-frequency plots for croaker revealed overlapping year-classes in the estuary, consistent with previous reports on croaker in South Carolina (Bearden, 1964; Miglarese *et al.*, 1982; Shealy *et al.*, 1974).

Bothidae - *Paralichthys dentatus* (summer flounder, Appendix VIII.A.20) and *P. lethostigma* (southern flounder, Appendix VIII.A.21), two important recreational species, were caught in low numbers throughout the year, with southern flounder increasing in abundance during March.

Soleidae - *Trinectes maculatus* (hogchoker) were collected year-round in the estuary. During Year 1, greatest abundances were observed in March, while in later years most were taken during summer months (July - September). Wenner *et al.* (1984) reported no apparent seasonality in hogchoker, while Ogburn *et al.* (1988) found peak abundance during summer. Hogchoker were consistently more abundant at the lower salinity stations (Appendix VIII.A.22). They were seldom caught in the Wando River, but were present in fairly high numbers at the uppermost Ashley River station, AR03.

Cynoglossidae - *Symphurus plagiusa* (blackcheek tonguefish) were collected every month with greatest abundances occurring in March and November (Appendix VIII.A.23). March collections generally contained adult individuals which later migrate seaward for spawning during May to October (Hildebrand and Cable, 1930). November collections included large numbers of newly recruited young-of-the-year. Wenner *et al.* (1984) found peak abundances only during the fall recruitment period. Although tonguefish were occasionally caught upriver, they were most abundant at higher salinity stations (CH02, CH03, WR01, AR01).

Community Analysis:

Normal cluster analysis of the data pooled by station and year indicated that station location was more significant than years in forming site groups (Table VIII.9). Five site groups were recognized in each season. While some exceptions occurred, the stations co-occurring were similar in each season. All collections from the freshwater station, CR04 (average salinity 0.0 ppt), formed one group. Tows from CR03, near the upriver transition zone between fresh and estuarine waters (ave. sal. 5.6 ppt), formed a second. Most collections from CR01 and WR03 clustered together as a third group.

Table VIII.9. Site groups resulting from normal cluster analyses by season based on data pooled by station and year.

SITE GROUPS

WINTER

Station	Year(s)
GROUP 1	
CR04	1 2 3 4
GROUP 2	
CR03	1 2 3 4
ARO3	4
GROUP 3	
WR02	1 2 4
WR03	1 4
GROUP 4	
CR01	2 3 4
CR02	4
WR01	4
WR03	2 3
AR01	4
GROUP 5	
CH02	1 2 3 4
CH03	2 3 4
CR01	1
CR02	1 2 3
WR01	1 2 3
WR02	3
AR02	4

SPRING

Station	Year(s)
GROUP 1	
CH02	1 2 3 4
CH03	2 3 4
CR01	1 2
CR02	1 2 3
WR01	2 4
AR01	4
AR02	4
GROUP 2	
CR02	4
WR01	1 3
WR02	1 2 3 4
GROUP 3	
CR01	3 4
WR03	1 2 3 4
GROUP 4	
CR04	1 2 3 4
GROUP 5	
CR03	1 2 3 4
AR03	4

SUMMER

Station	Year(s)
GROUP 1	
CR04	1 2 3 4
GROUP 2	
CH02	2 3 4
CH03	2 3 4
CR02	2 4
WR01	2 3 4
WR02	4
AR01	4
GROUP 3	
CH02	1
CH03	1
CR02	3
WR01	1
WR02	1 2 3
GROUP 4	
CR01	1 2 3 4
WR03	1 2 3 4
GROUP 5	
CR02	1
CR03	1 2 3 4
AR03	4

FALL

Station	Year(s)
GROUP 1	
CR03	1
CR04	1 2 3 4
GROUP 2	
CR01	1 2 3
WR03	2 3 4
GROUP 3	
CR03	2
WR02	1 2 3
WR03	1
GROUP 4	
CH02	1 2 3 4
CH03	2 3 4
CR01	4
CR02	2 3
WR01	1
AR01	4
GROUP 5	
CR02	1 4
CR03	3 4
WR01	2 3 4
WR02	4
AR02	4
AR03	4

Composition of the two remaining groups varied seasonally. Summer pre-rediversion trawls from remaining stations formed a fourth group while post-rediversion tows from these stations formed a fifth. In winter, most collections from the two upriver stations in the Wando formed the fourth group with remaining stations grouping together for the fifth. In spring, catches from WR02, with several other years and stations were grouped in a fourth category with most years from lower stations forming the fifth group. In fall, the CR03 group was not distinct, while collections from most lower river stations in Year 4 formed a group. Also, the two upriver Wando stations clustered together for a fourth group, and most harbor stations formed the last group.

For all four seasons, seven to eleven groups were formed by inverse (species) cluster analyses (Table VIII.10). Consistently present throughout the year was a group of very abundant and ubiquitous species, including *Leiostomus xanthurus, Micropogonias undulatus, Stellifer lanceolatus, Anchoa mitchelli, Symphurus plagiusa*, and *Callinectes sapidus*. This group (A) exhibited the greatest separation from other species groups. It ranked high in constancy and low in fidelity over most site groups in the nodal analysis. (Figures VIII.10 - VIII.13). Thus, this assemblage was distinct, it was present throughout the estuary except at the freshwater station, and it was represented by several species that were dominant in the catches throughout the year in all years.

Another group consistently present was formed by primarily freshwater species such as *Ictalurus catus, I. furcatus, I. punctatus*, and *Morone americana*. High constancy and fidelity were indicated for these with the freshwater (CR04) site group. In winter, lower fidelity and low constancy in all site groups indicated that some of these species occurred at other stations throughout the estuary.

For each season, other groups were high in constancy or fidelity. In general, a group of widely occurring species with low abundance demonstrated high constancy or low fidelity in each season (group I in winter, C in spring, D and I in summer, H in fall). Another assemblage occurring in most seasons ranked high in constancy and fidelity for the transition zone between fresh and estuarine waters (CR03). This assemblage occurred in all seasons except fall, though its composition changed seasonally (B and C in winter, F in spring, J in summer). Overall, prominent stratification occurred at the freshwater station (CR04) and the nearby transition zone (CR03). The remaining stations sampled in the Charleston Harbor system were not consistently stratified during the four seasons.

Community diversity (H') fluctuated with both time and location, typically being lowest in winter (0.3) and highest in summer (4.2) (Figures VIII.14-17). There were inconsistent yearly fluctuations in diversity among the stations except in fall when Year 4

Table VIII.10. Species groups resulting from inverse cluster analyses by season based on data pooled by station and year.

SPECIES GROUPS

WINTER

GROUP A

Anchoa [illegible]
Leiostomus xanthurus
Brevoortia tyrannus
Micropogonias undulatus
Stellifer lanceolatus
Penaeus setiferus
Symphurus plagiusa
Urophycis regia
Palaemonetes vulgaris
Callinectes sapidus
Paralichthys lethostigma

GROUP B

Morone americana
Ictalurus furcatus
Trinectes maculatus
Ictalurus catus
Rhithropanopeus harrisii
Anguilla rostrata
Dorosoma petenense
Ictalurus punctatus

GROUP C

Pomatomus saltatrix
Morone saxatilis
Lepisosteus osseus

GROUP D

Callinectes similis
Limulus polyphemus
Scophthalmus aquosus
Periclimenes longicaudatus
Libinia emarginata

GROUP E

Portunus gibbesii
Squilla empusa
Portunus spinimanus
Etropus crossotus
Trachypenaeus constrictus
Menticirrhus americanus
Pagurus longicarpus
Lolliguncula brevis
Libinia dubia
Neopanope sayi
Cancer irroratus
Menidia beryllina

GROUP F

Centropristis striata
Hypsoblennius hentzi
Alosa sapidissima
Alosa aestivalis
Xanthidae

SPRING

GROUP A

Leiostomus xanthurus
Micropogonias undulatus
Anchoa mitchilli
Urophycis regia
Lolliguncula brevis
Callinectes sapidus
Symphurus plagiusa
Trachypenaeus constrictus
Penaeus duorarum
Stellifer lanceolatus

GROUP B

Opsanus tau
Panopeus herbstii
Neopanope sayi
Urophycis floridanas
Libinia dubia
Bairdiella chrysoura

GROUP C

Paralichthys lethostigma
Rhithropanopeus harrisii
Dasyatis sabina
Pomatomus saltatrix
Brevoortia tyrannus
Trinectes maculatus
Ariopsis felis
Palaemonetes vulgaris
Cynoscion regalis
Trichiurus lepturus
Penaeus setiferus
Peprilus triacanthus
Ophidion marginata
Scophthalmus aquosus
Ancylopsetta quadrocellata
Paralichthys dentatus
Prionotus tribulus

GROUP D

Cynoscion nothus
Anchoa hepsetus
Pagurus longicarpus
Ovalipes ocellata

GROUP E

Etropus crossotus
Squilla empusa
Cancer irroratus
Portunus gibbesii
Centropristis striata
Centropristis philadelphica
Callinectes similis
Ophichthus gomesi
Clibanarius vittatus

SUMMER

GROUP A

Leiostomus xanthurus
Micropogonias undulatus
Cynoscion regalis
Penaeus aztecus
Anchoa mitchilli
Stellifer lanceolatus
Penaeus setiferus
Symphurus plagiusa
Callinectes sapidus
Menticirrhus americanus
Trachypenaeus constrictus
Lolliguncula brevis
Ariopsis felis
Bairdiella chrysoura
Palaemonetes vulgaris
Chaetodipterus faber
Penaeus duorarum
Peprilus alepidotus

GROUP B

Ictalurus furcatus
Ictalurus punctatus
Ictalurus catus
Trinectes maculatus
Macrobrachium ohione

GROUP C

Periclimenes longicaudatus
Latreutes parvulus
Xanthidae
Acetes americanus

GROUP D

Callinectes similis
Squilla empusa
Pagurus longicarpus
Portunus gibbesii
Centropristis philadelphica
Etropus crossotus
Anchoa hepsetus
Bagre marinus
Brevoortia tyrannus
Paralichthys lethostigma
Chloroscombrus chrysurus
Libinia dubia
Neopanope sayi
Clibanarius vittatus
Ophichthus gomesi
Rhizoprionodon terraenovae
Trichiurus lepturus
Paralichthys dentatus
Selene vomer
Synodus foetens
Citharichthys spilopterus
Ophidion marginata

FALL

GROUP A

Symphurus plagiusa
Trachypenaeus constrictus
Callinectes similis
Leiostomus xanthurus
Callinectes sapidus
Micropogonias undulatus
Bairdiella chrysoura
Cynoscion regalis
Lolliguncula brevis
Penaeus duorarum
Stellifer lanceolatus
Penaeus setiferus
Anchoa mitchilli
Prionotus tribulus
Squilla empusa
Menticirrhus americanus
Paralichthys lethostigma
Centropristis philadelphica
Etropus crossotus

GROUP B

Ictalurus catus
Ictalurus furcatus
Ictalurus punctatus
Morone americana

GROUP C

Pagurus longicarpus
Portunus gibbesii
Ophidion marginata
Dasyatis sabina

GROUP D

Ophichthus gomesi
Larimus fasciatus
Lysmata wurdemanni
Caranx hippos
Chaetodipterus fabe

GROUP E

Archosargus probatocephalus
Eurypanopeus depressus
Hypsoblennius hentzi
Lagodon rhomboides
Lutjanus griseus

GROUP F

Brevoortia tyrannus
Peprilus alepidotus
Selene vomer
Selene setapinnis
Paralichthys dentatus
Trinectes maculatus
Rhithropanopeus harrisii

Table VIII.10. Continued:

GROUP G

Astroscopus y-graecum
Myrophis punctatus
Clibanarius vittatus

GROUP H

Gobiesox strumosus
Prionotus scitulus
Gobiosoma bosci
Eurypanopeus depressus

GROUP I

Cynoscion regalis
Penaeus duorarum
Centropristis philadelphica
Urophycis floridana
Ancylopsetta quadrocellata
Paralichthys dentatus
Prionotus tribulus
Lagodon rhomboides
Cynoscion nebulosus
Palaemonetes pugio
Mugil cephalus
Opsanus tau
Panopeus herbstii
Bairdiella chrysoura
Alpheus heterochaelis

GROUP J

Sciaenops ocellata
Dorosoma cepedianum

GROUP K

Ophidion marginata
Peprilus alepidotus
Mugil curema

GROUP F

Ictalurus catus
Ictalurus furcatus
Morone americana
Macrobrachium ohione
Anquilla rostrata

GROUP G

Gobiesox strumosus
Eurypanopeus depressus
Xanthidae
Syngnathus louisianae

GROUP E

Chilomycterus schoepfi
Sphoeroides maculatus
Syngnathus fuscus
Prionotus tribulus
Syngnathus louisianae
Scomberomorus maculatus
Selene setapinnis

GROUP F

Larimus fasciatus
Dasyatis sayi
Monacanthus hispidus
Leptochela serratorbita
Centropristis striata
Hypsoblennius hentzi
Menippe mercenaria
Prionotus scitulus
Orthopristis chrysoptera

GROUP G

Alphaeus heterochaelis
Opisthonema oglinum

GROUP H

Cynoscion nebulosus
Sphyraena guachancho
Limulus polyphemus

GROUP I

Opsanus tau
Panopeus herbstii
Dasyatis sabina
Pomatomus saltatrix
Lagodon rhomboides
Eurypanopeus depressus

GROUP J

Rhithropanopeus harrisii
Palaemonetes pugio
Dorosoma petenense

GROUP K

Caranx hippos
Archosargus probatocephalus

GROUP G

Synodus foetens
Chloroscombrus chrysurus
Pomatomus saltatrix

GROUP H

Opsanus tau
Panopeus herbstii
Palaemonetes vulgaris
Anchoa hepsetus
Periclimenes longicaudatus

GROUP I

Syngnathus louisianae
Cynoscion nebulosus

GROUP J

Citharichthys spilopterus
Penaeus aztecus
Neopanope sayi
Libinia dubia
Alpheus heterochaelis

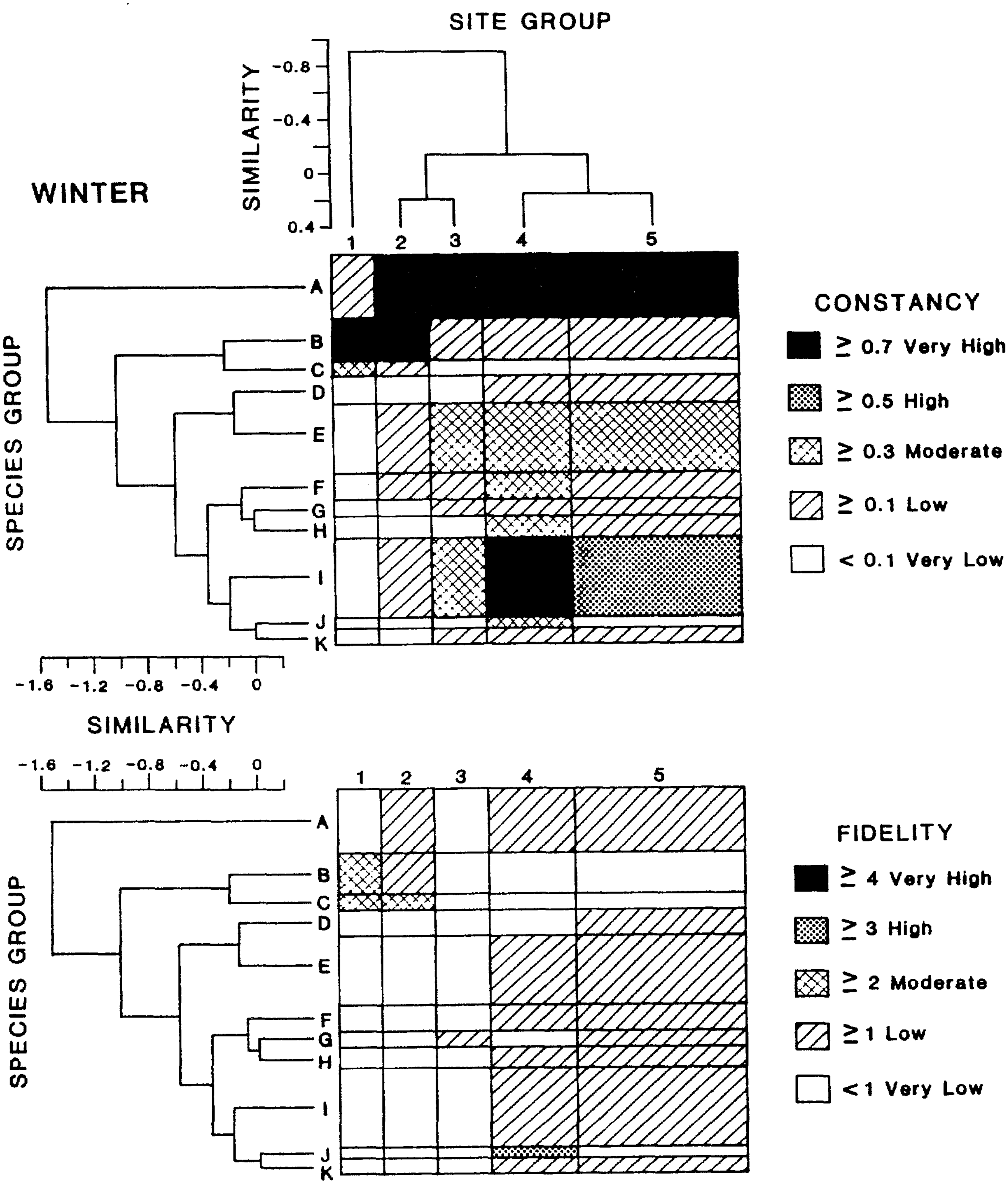


Figure VIII.10. Constancy and fidelity tables from nodal analysis of winter data pooled by station and year.

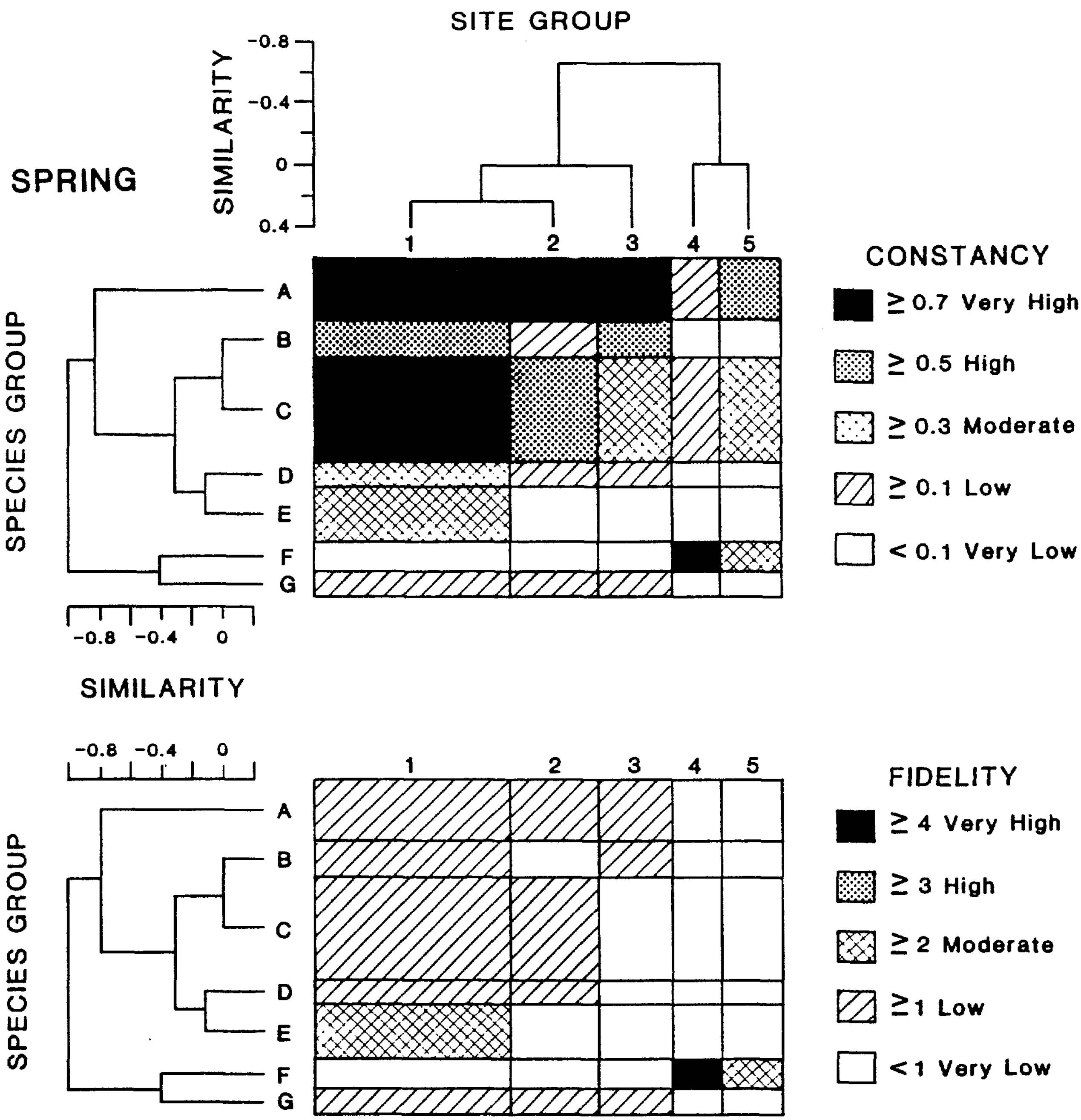


Figure VIII.11. Constancy and fidelity tables from nodal analysis of spring data pooled by station and year.

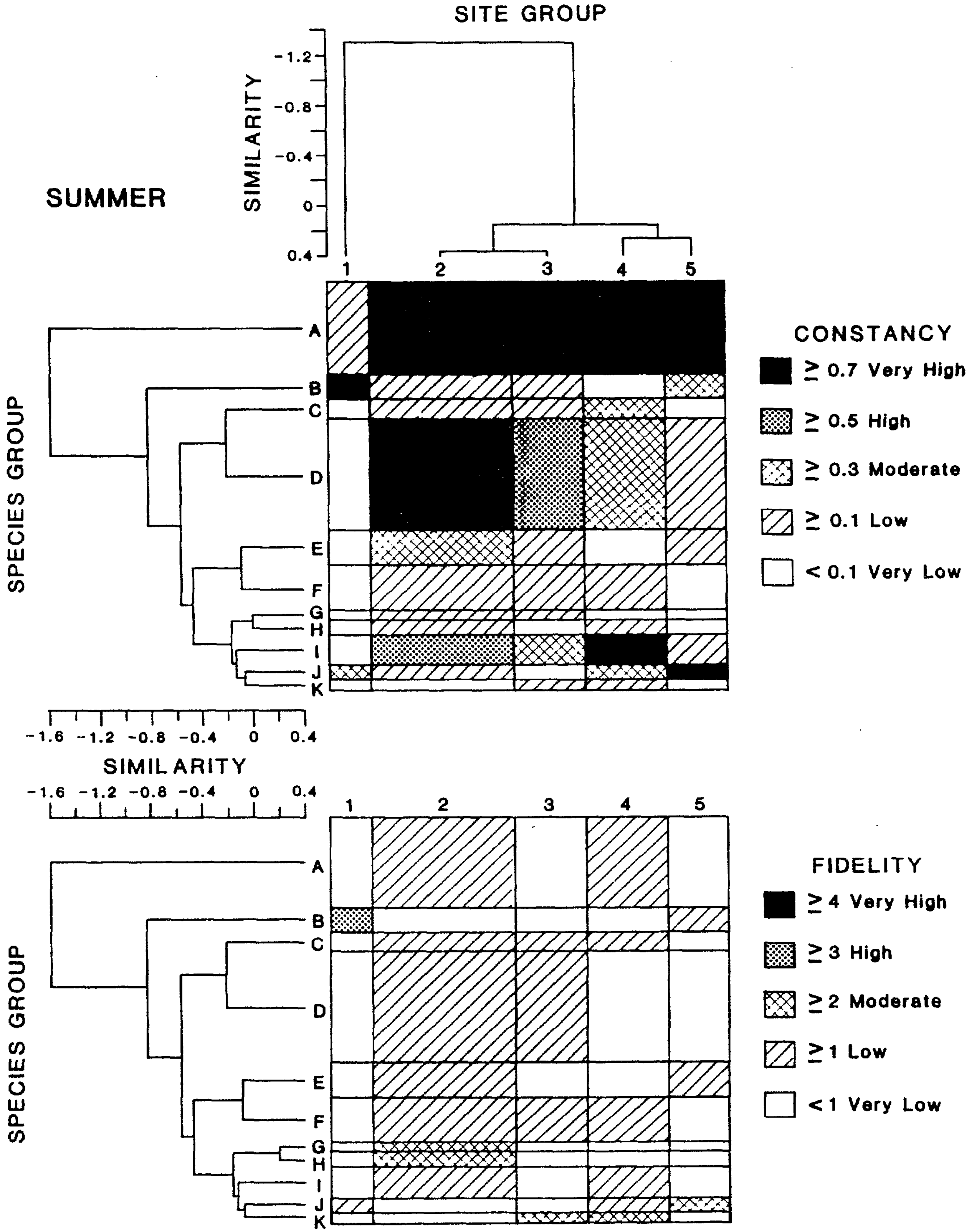


Figure VIII.12. Constancy and fidelity tables from nodal analysis of summer data pooled by station and year.

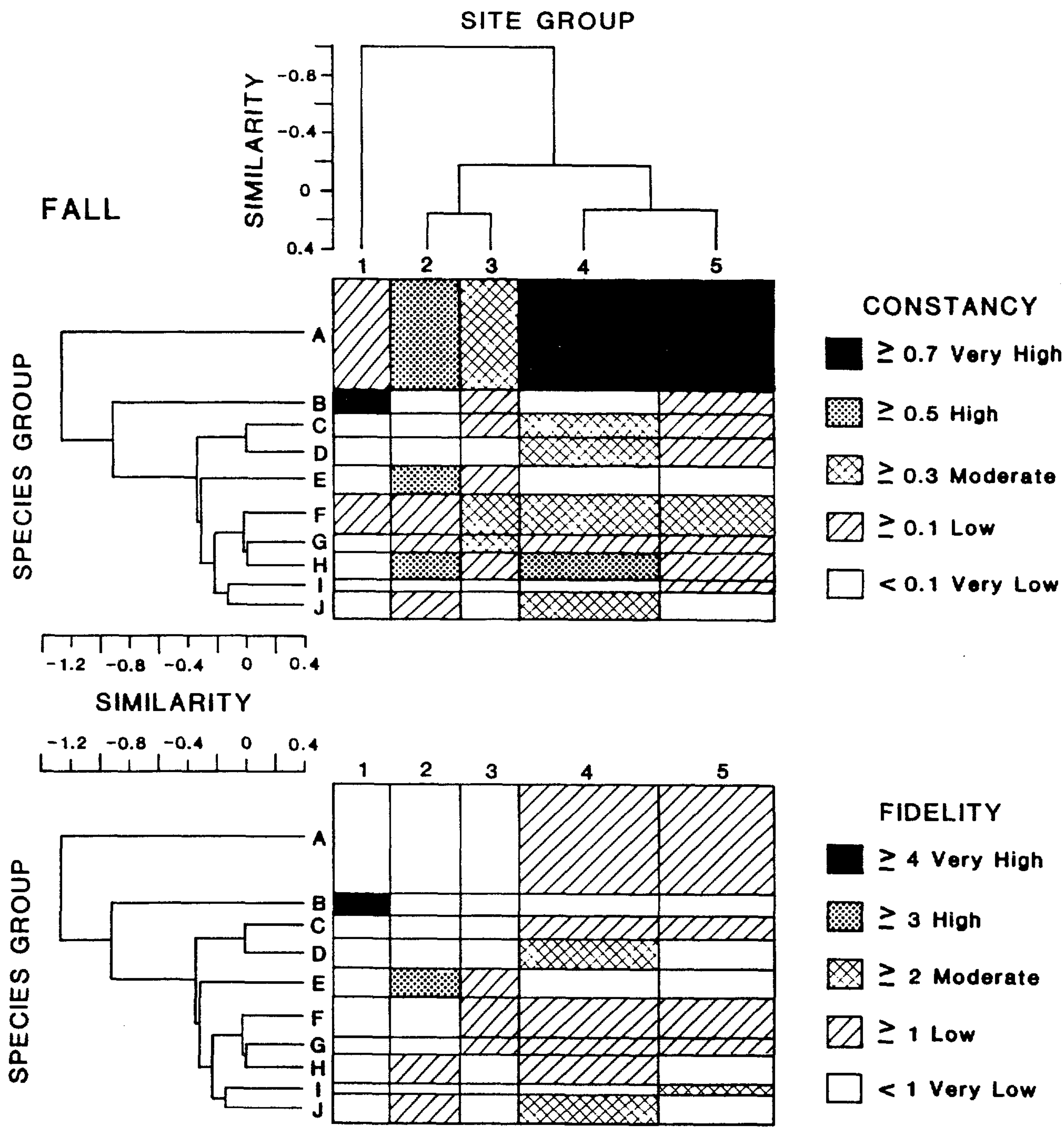


Figure VIII.13. Constancy and fidelity tables from nodal analysis of fall data pooled by station and year.

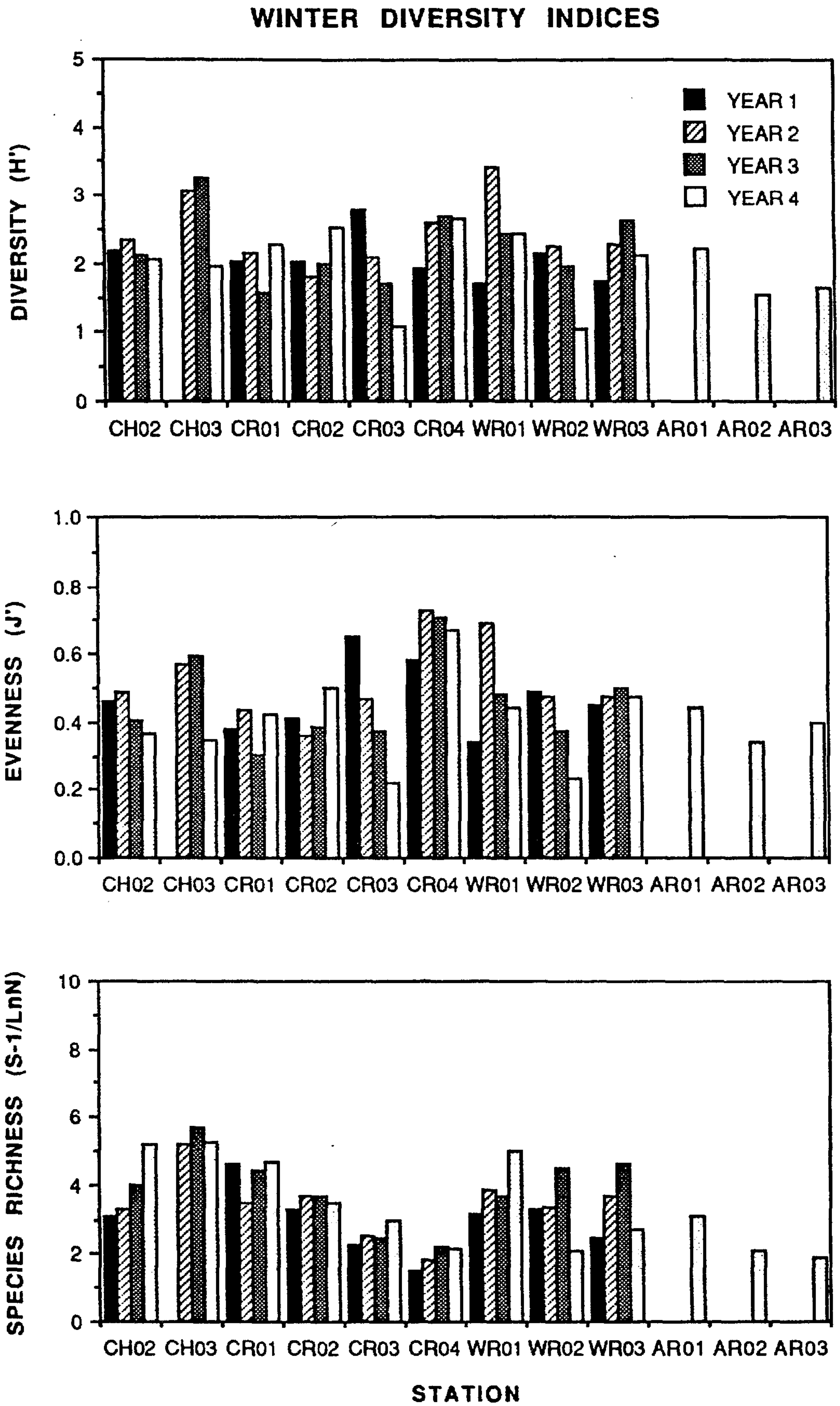


Figure VIII.14. Winter diversity, evenness, and species richness indices by station for each year.

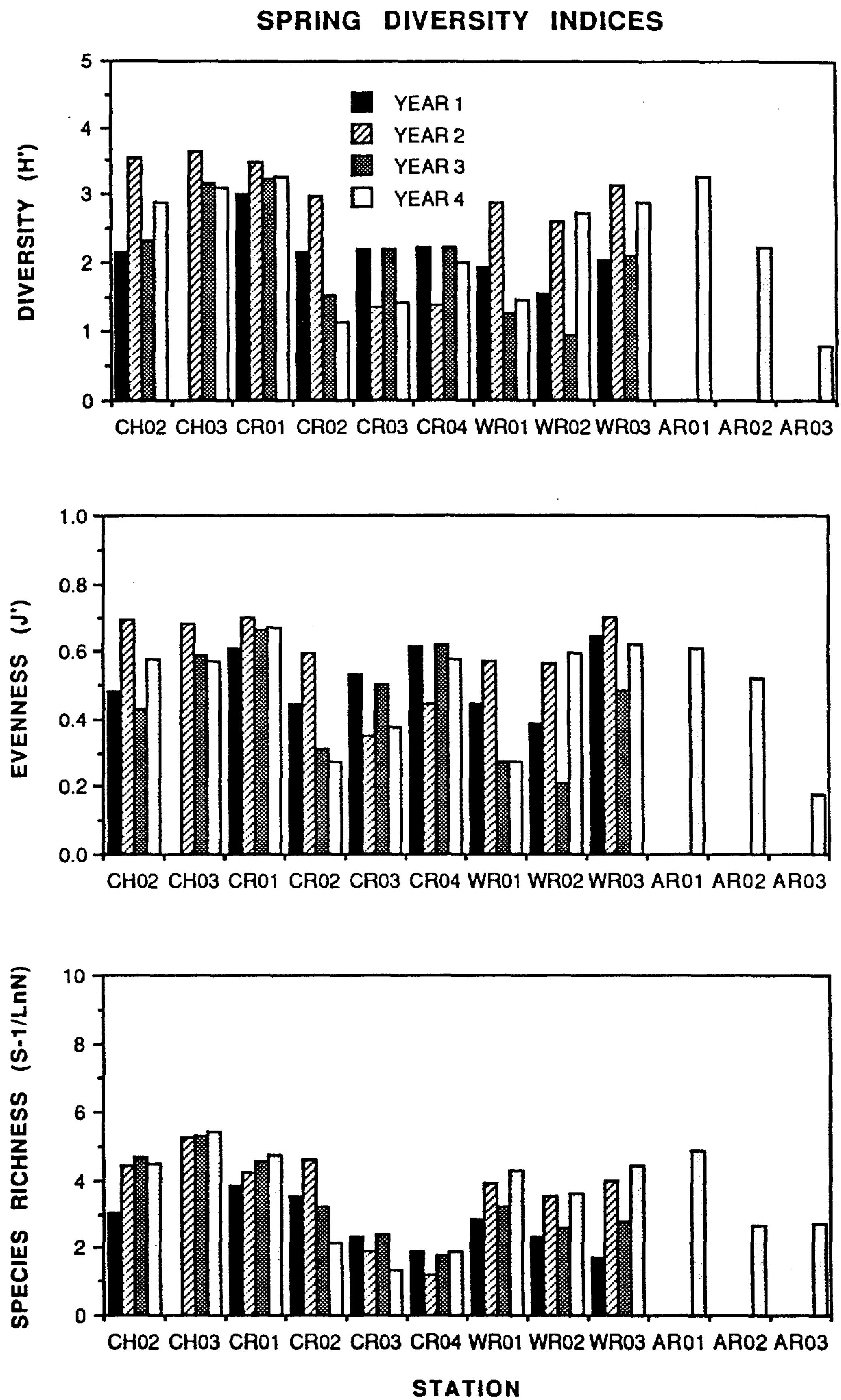


Figure VIII.15. Spring diversity, evenness, and species richness indices by station for each year.

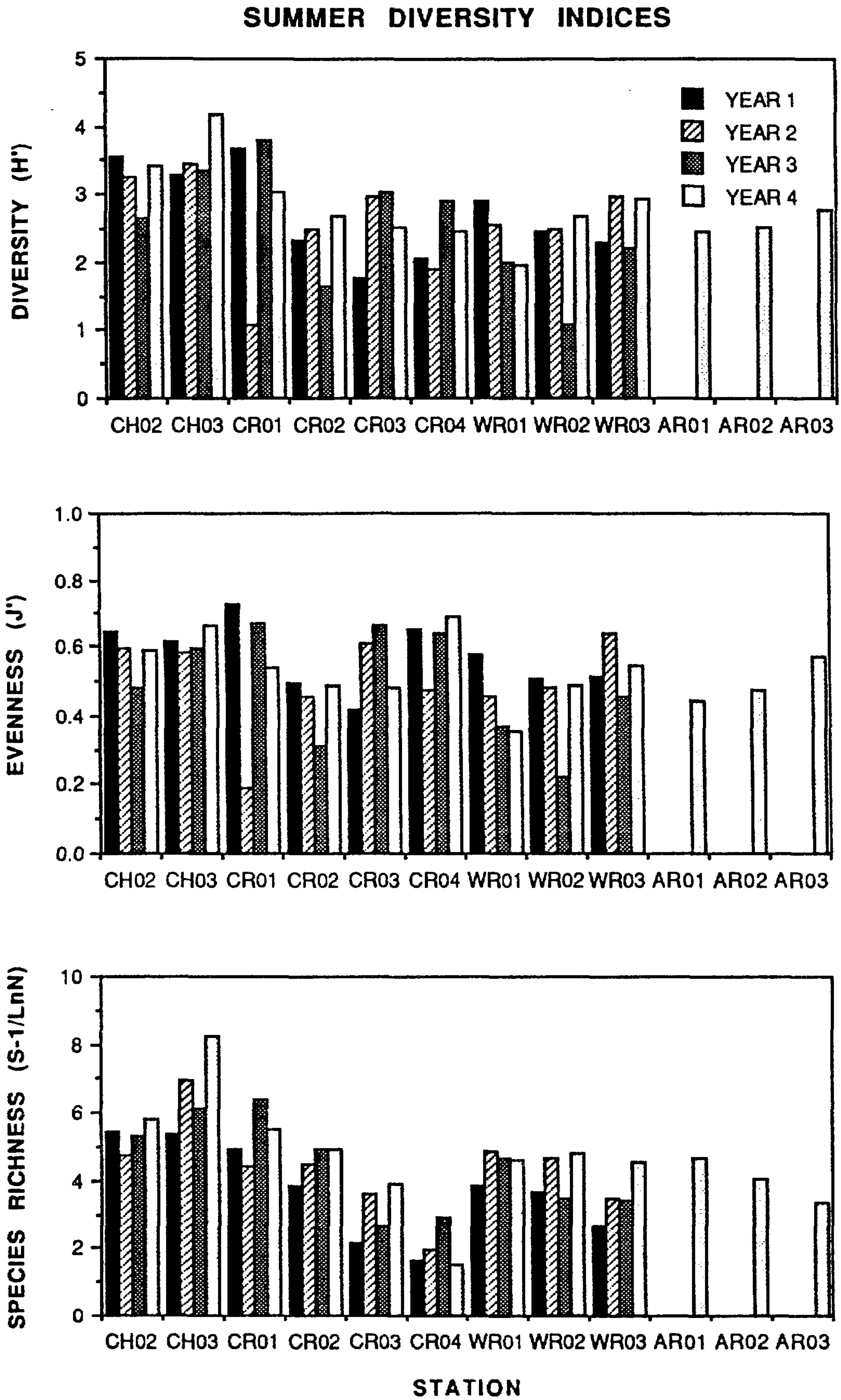


Figure VIII.16. Summer diversity, evenness, and species richness indices by station for each year.

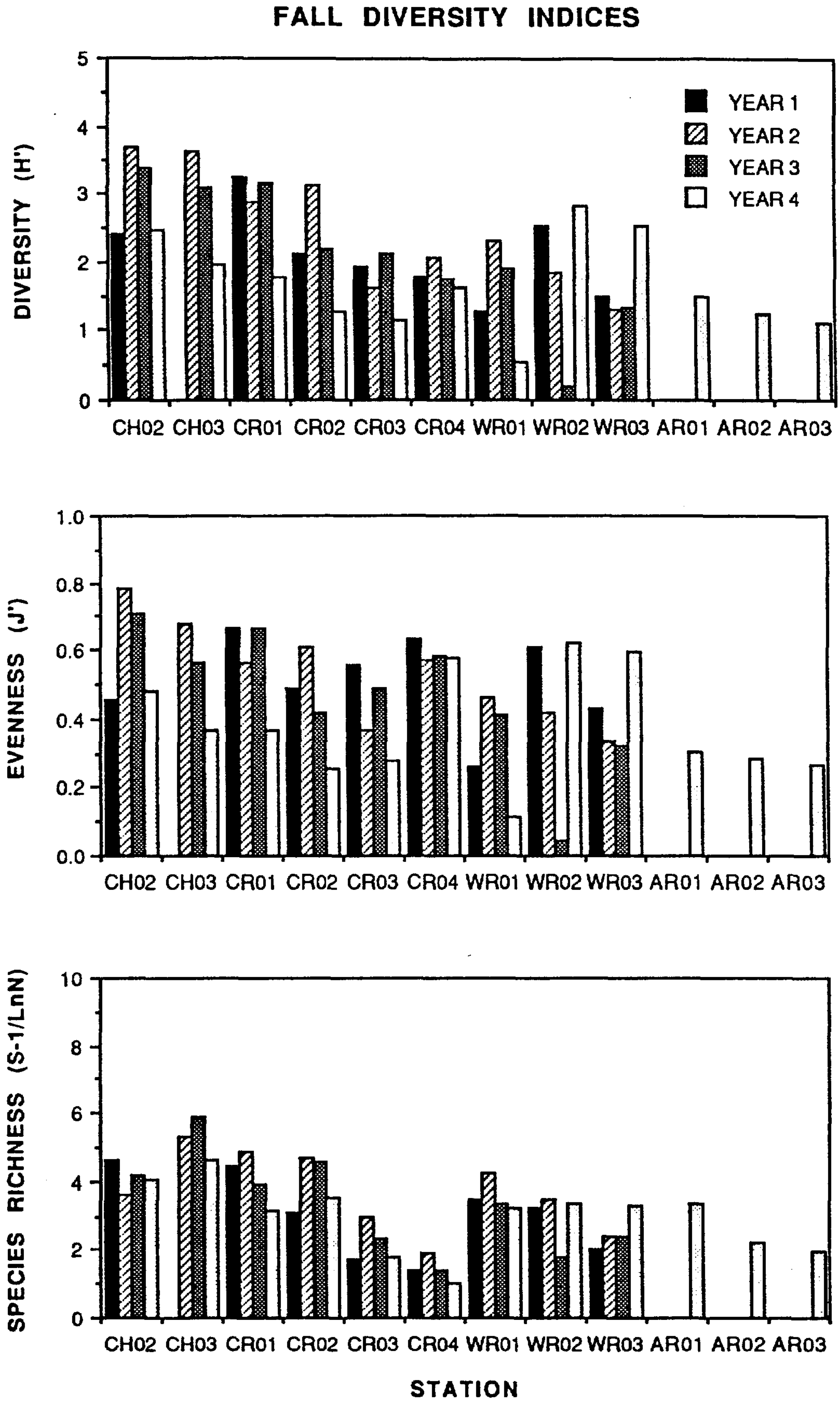


Figure VIII.17. Fall diversity, evenness, and species richness indices by station for each year.

was typically very low in diversity. The seasonal trends in diversity found in this project were similar to trends noted in other studies. However, the maximal summer values found in this study were often twice those calculated for other estuaries in Georgia (Dahlberg and Odum, 1970), Maryland (McErlean *et al.*, 1973), Massachusetts (Haedrich and Haedrich, 1974), and South Carolina (Cain and Dean, 1976). Similar summer diversity indices occurred only in coastal waters off the southeastern United States (Wenner and Sedberry, 1989), reflecting an abundant and rich fauna available as a source for immigration into the harbor. The upriver decrease in diversity differed from studies by Dahlberg and Odum (1970) and McErlean *et al.* (1973). Increased utilization of the harbor and lower rivers by more typically coastal species could have explained this observation.

Evenness was generally highest in summer and spring and lowest in winter and fall, similar to the results of McErlean *et al.* (1973) in Maryland. Evenness was highest at the upriver and mouth stations in the Cooper River and in the harbor. Elsewhere, evenness increased upriver similar to trends noted in other studies (Dahlberg and Odum, 1970; McErlean *et al.*, 1973).

Species richness was highest in summer and remained fairly constant during the other seasons. Again, peak summer richness indices were more than twice that reported for other areas except in coastal waters along the southeastern United States. Similarity between the estuary and coastal habitats in both richness and diversity was in contrast to the sharp difference Dahlberg and Odum (1970) found in Georgia.

Throughout this chapter, several changes have been discussed comparing current conditions in the Charleston Harbor estuarine system to previous conditions; however, many of these changes are atypical of an estuary whose fresh water inflow has been reduced. In a typical estuary, the oligohaline zone is an important nursery area for new recruits. Many species utilize the shallows of these areas independent of salinity (Rozas and Hackney, 1984). Many species utilize tidal stream transport to initially colonize this upper area of the estuary. Increased flow rates displace the freshwater line seaward, compress this freshwater boundary horizontally and vertically, and prevent flood-tide displacement into the oligohaline recruitment areas (Rogers *et al.*, 1984). Hence, a decrease in flowrate, as occurred in this rediversion, should enhance this recruitment process. However, Benson (1981) suggests that reductions of flowrates by diversions result in a reduction in the overall size of the estuarine nursery habitat and in disruption of spawning and nursery cycles that are closely associated with peak spring runoff. A reduction of flow by as little as 30-40% can destroy the dynamic equilibrium of an estuary within three to seven years and may increase the impact of pollutants by four to twelve times (Hedpeth and Rozengurt, 1985). Reduced flowrates may cause losses in commercial fisheries as occurred in Texas on menhaden, white shrimp, and brown shrimp (Funicelli, 1984).

In many ways, the Charleston Harbor estuarine system is a typical estuary in its role in recruitment and as a nursery area. However, instead of the losses and destruction reported elsewhere, there has been an apparent increase in the use of this estuary by many more species as a nursery area, particularly considering that this study segment sampled only the main channels of the rivers and not the smaller creeks. It is possible that coincidental environmental conditions, *i.e.* drought or cold winters, may have caused any negative effects of a diversion to be eliminated, masked, or postponed. It may be that the continued regulation of the flow as opposed to absolute elimination has contributed to an improved end result. However, another possibility is that changes are occurring on the order of seven instead of three years and that the current results represent a transitional phase in this process, perhaps through the loss of the adults and the reduction of the diversity of the habitat to that of only a temporary nursery ground where pollutants and other environmental changes may seriously impact the existing population. There are not sufficient data in the present study to resolve which, if any, of these possibilities are occurring.

SUMMARY

During the four-year study, several changes were identified:

1. There was a significant increase in the number of species of fish and invertebrates occurring within the Charleston Harbor estuarine system. This change coincided with the rediversion of the flow of the Cooper River. Concurrent with this increase in the number of species, there was a general decline in the occurrence of pre-rediversion species.

2. The number of individuals collected per trawl tow significantly increased for fish and invertebrates during the study period.

3. Although there was no significant change in biomass per tow for all taxa combined, there was a significant decrease in fish biomass and a significant increase in the biomass of invertebrates over the study period.

4. In both abundance and biomass, the dominant species of fish collected prior to rediversion in this and previous studies remained the dominant species following rediversion.

5. For invertebrates, most taxa that were previously dominant in abundance remained abundant after rediversion with three exceptions. One previously abundant species was not found in collections after rediversion, while two taxa currently dominant were not identified from collections before rediversion.

6. Of the 23 species analyzed for differences in abundance, biomass, and distribution:
   a. there were no differences found in abundance or biomass in 5 species:
      Lolliguncula brevis (brief squid)
      Penaeus aztecus (brown shrimp)
      Ictalurus furcatus (blue catfish)
      Paralichthys dentatus (summer flounder)
      Symphurus plagiusa (blackcheek tonguefish)

   b. eleven species exhibited variable year to year differences:
      Penaeus duorarum (pink shrimp)
      Trachypenaeus constrictus (roughneck shrimp)
      Callinectes similis (lesser blue crab)
      Brevoortia tyrannus (Atlantic menhaden)
      Ictalurus catus (white catfish)
      Ariopsis felis (sea catfish)
      Urophycis regia (spotted hake)
      Bairdiella chrysoura (silver perch)
      Cynoscion regalis (weakfish)
      Stellifer lanceolatus (star drum)
      Trinectes maculatus (hogchoker)

   c. four species increased in abundance after rediversion:
      Penaeus setiferus (white shrimp)
      Palaemonetes vulgaris (grass shrimp)
      Anchoa mitchilli (bay anchovy)
      Paralichthys lethostigma (southern flounder)

   d. a decrease in abundance after rediversion occurred in one species:
      Callinectes sapidus (blue crab)

e. two species increased in number but decreased in size and/or weight indicating a greater utilization of the estuary as a nursery habitat after rediversion:
   Leiostomus xanthurus (spot)
   Micropogonias undulatus (Atlantic croaker)

f. an additional eight species showed an increase in the percentage of smaller individuals:
   Penaeus setiferus (white shrimp)
   Anchoa mitchilli (bay anchovy)
   Ictalurus catus (white catfish)
   Ictalurus furcatus (blue catfish)
   Cynoscion regalis (weakfish)
   Stellifer lanceolatus (star drum)
   Paralichthys dentatus (summer flounder)
   Paralichthys lethostigma (southern flounder)

g. nine species had peak abundances 11 - 22 km (6 - 12 miles) further upriver after rediversion:
   Penaeus duorarum (pink shrimp)
   Penaeus setiferus (white shrimp)
   Callinectes sapidus (blue crab)
   Anchoa mitchilli (bay anchovy)
   Ictalurus catus (white catfish)
   Ictalurus furcatus (blue catfish)
   Leiostomus xanthurus (spot)
   Micropogonias undulatus (Atlantic croaker)
   Trinectes maculatus (hogchoker).

In conclusion, while the species composition of the finfish and invertebrate communities has remained similar over the past four years, consistent increases in abundance and decreases in size and biomass of several dominant species confirmed the increased use of this estuarine system as a nursery ground and its decreased use as a habitat for adults.